Condition and Trends Assessment

Chapter 3. Analytical Approaches for assessing Ecosystem Condition and Human Well-Being

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Main Messages

- **A variety of tools are available to assess ecosystem condition and support policy decisions that involve trade-offs among ecosystem services.** For example, clearing forested land affects multiple ecosystem services (e.g., food production, biodiversity, and watershed protection), each of which affects human well-being (e.g., increased income from crops, reduced tourism value of biodiversity, and damage from downstream flooding). Assessing these trade-offs in the decision-making process requires scientifically based analysis to quantify the responses to different management alternatives. Scientific advances over the past few decades, particularly in computer modeling, remote sensing, and environmental economics, make it possible to assess these linkages.

- **The availability and accuracy of data sources and methods for this assessment are unevenly distributed for different ecosystem services and geographic regions.** Data on provisioning services, such as crop yield and timber production, are usually available. On the other hand, data on regulating, supporting, and cultural services such as nutrient cycling, climate regulation, or aesthetic value are seldom available, making it necessary to use indicators, model results, or extrapolations from case studies as proxies. Methods for quantifying ecosystem responses are also uneven. Methods to estimate crop yield responses to fertilizer application, for example, are well developed. But methods to quantify relationships between ecosystem services and human well-being, such as the effects of deteriorating biodiversity on human disease, are at an earlier stage of development.

- **Ecosystems respond to management changes on a range of time and space scales, and careful definition of the scales included in analyses is critical.** Soil nutrient depletion, for example, occurs over time-scales of decades and would not be captured in an analysis based on a shorter time period. Some of the impact of deforestation is felt in reduced water quality far downstream; an analysis that only considers the forest area itself would miss this impact. Ideally, analysis at varying scales would be carried out to properly assess trade-offs. In particular, it is essential to consider non-linear responses of ecosystems to perturbations in analysis of trade-offs, such as loss of resilience to climate variability below a threshold number of plant species.

- **Ecosystem condition is only one of many factors that affect human well-being, making it challenging to assess linkages between them.** Health outcomes, for example, are the combined result of ecosystem condition, access to health care, economic status, and myriad other factors. Interpretations of trends in indicators of well-being must appropriately account for the full range of factors involved. The impacts of ecosystem change on well-being are often subtle, which is not to say unimportant; impacts need not be drastic to be significant. A small increase in food prices resulting from lower yields will affect many people, even if none starve as a result. Tracing these impacts is often difficult, particularly in aggregate analyses where the signal of the effect of ecosystem change is often hidden by multiple confounding factors. Analyses linking well-being and ecosystem condition are most easily carried out at a local scale, where the linkages can be most clearly identified.

- **Ultimately, decisions about trade-offs in ecosystem services require balancing societal objectives, including utilitarian and non-utilitarian objectives, short and long term objectives, and local-scale and global-scale objectives.** The analytical approach for this report aims to quantify, to the degree possible, the most important trade-offs within different ecosystems and among the ecosystem services, as input to weigh societal objectives based on comprehensive analysis of the full suite of ecosystem services.

3.1 Introduction

The Conditions and Trends report systematically assesses the current state and recent trends in the world's ecosystems and their services and the significance of these changes for human livelihoods, health, and well-being. The individual chapters in the report draw on a wide variety of data sources and analytical methods from both the natural and social sciences. This chapter provides an overview of many of these data and methods, their basis in the scientific literature, and the limitations and possibilities for application to the assessment of ecosystem conditions and trends.

The data and methods used throughout the report provide the foundation for assessing linkages between management decisions and other drivers of ecosystem change, trends in ecosystem services, and implications for human well-being ({Figure 3.1}). The MA's approach is premised on the notion that management decisions generally involve trade-offs among ecosystem services, and that quantitative and scientifically-based assessment of the trade-offs is a necessary ingredient for sound decision-making. For example, decisions to clear land for agriculture involve trade-offs between food production and protection of biological resources; decisions to extract timber involve trade-offs between income from timber sales and watershed protection; and decisions to designate marine protected areas involve trade-offs between preserving fish stocks and availability of fish or jobs for local populations. Accounting for these trade-offs involves quantifying the effects of the management decision on ecosystem services and human-well being in comparable units over varying spatial and temporal scales.

{Section 3.2} of this chapter discusses the data and methods used in the chapters to assess conditions and trends in ecosystems and their services. Individual chapters of this report apply these methods to identify the implications of changes in ecosystem condition (e.g., forest conversion to cropland) for ecosystem services (e.g., flood protection). Rigorous analyses of these linkages are a key prerequisite to quantifying the effects on human well-being (e.g., damage from downstream flooding).

Figure 3.1: Linking ecosystem condition to well-being.

{Section 3.3} discusses data and methods for quantifying the effects of changes in ecosystem services on human well-being, including human health, economic costs and benefits, poverty and other measures of well-being, and on the intrinsic value of ecosystems. These methods provide a framework for assessing management decisions or policies that alter ecosystems, based on comprehensive information about the repercussions for human well-being from intentional or unintentional alteration of ecosystem services. {Section 3.4} discusses approaches for assessing trade-offs from management decisions. These approaches aim to quantify, in comparable units, the repercussions of a decision for the full range of ecosystem services. The approaches must also account for the varying spatial and temporal scale over which management decisions alter ecosystem services. Decisions to clear forests, for example, provide immediate economic benefits for local interests but contribute to the increase of greenhouse gases in the atmosphere with longer-term implications at the global scale.

This chapter provides a general overview of the available methods and data sources and their applicability to the assessment. For detailed descriptions of data sources used in reference to a particular ecosystem or service, we refer the reader to individual chapters. Core data sets used by all chapters to ensure consistency and comparability among the different ecosystems are described in {Appendix 1}.

3.2 Assessing Ecosystem Conditions and Trends

The foundation for analyses carried out in individual chapters of this report is basic information about each ecosystem service (Part II chapters) and spatially-defined ecosystem (Part III chapters). To greater and lesser degrees, each chapter assesses the following basic information and derives conclusions about the important trends in ecosystem condition and trade-offs among ecosystem services. The chapters apply various methods to assess the significance of these trends for human well-being (see {section 3.3}). The basic information serving as the foundation from which to assess conditions and trends in ecosystems includes:

- What are the current spatial extent and condition of ecosystems?
- What are the quality, quantity, and spatial distributions of services provided by the systems?
- Who lives in the ecosystem and what ecosystem services do they use?
- What are the trends in ecosystem condition and their services in the recent (decades) and more distant past (centuries)?
- How do ecosystem condition, and in turn ecosystem services, respond to the drivers for each system?

Table 3.1: Data sources and analytical approaches for assessing ecosystem conditions and trends

The availability of data and applicability of methods to derive this basic information ({Table 3.1}) vary from ecosystem to ecosystem, service to service, and even region to region within an ecosystem. For example, the United Nations Food and Agriculture Organization (FAO) reports data on agricultural products, timber, and fisheries at the national level (e.g., (FAO 2000a). Although data reliability is sometimes questionable due to known problems such as definitions that vary between data-submitting countries (see {Section 3.2.2}), such data on “provisioning” ecosystem services with value as commodities are generally available. On the other hand, data on the spatial distribution, quantity, and quality of “regulating”, “supporting”, and “cultural” services such as nutrient cycling, climate regulation, or aesthetic value have generally not been collected and it is necessary to use indicators, modeled results, or extrapolations from case studies as proxy data. Within a given ecosystem service or geographic system, resource inventories and census data are generally more readily available and reliable in developed than in developing countries.

The following sections provide overviews of each of these data sources and analytical approaches used throughout the report.

3.2.1 Remote Sensing and Geographic Information Systems

The availability of data to monitor ecosystems on a global scale is the underpinning for the MA. Advances in remote sensing technologies over the past few decades now enable repeated observations of the Earth's surface. The potential to apply these data for assessing trends in ecosystem condition is only beginning to be realized. Moreover, advances in analytical tools such as Geographic Information Systems (GIS) allow data on the physical, biological, and socioeconomic characteristics of ecosystems to be assembled and interpreted in a spatial framework, making it feasible to establish linkages between drivers of change and trends in ecosystem services.

3.2.1.1 Remote Sensing

Ground-based surveys for mapping vegetation and other biophysical characteristics can be carried out over limited areas but it would be an enormous undertaking to carry out globally comprehensive ground-based surveys over the entire surface of the Earth. Remote sensing, broadly defined as the science of obtaining information about an object without being in direct physical contact (Colwell 1983), is the primary data source for mapping the extent and condition of ecosystems over large areas. Moreover, remote sensing provides measurements that are consistent over the entire area being observed and are not subject to varying data collection methods in different locations as are ground-based measurements. Repeated observations using the same remote sensing instrument also provide measurements that are consistent through time as well as through space.

Most remote sensing data useful to assess ecosystem conditions and trends are obtained from sensors on satellites ({Table 3.2}). Satellite data are generally digital and consequently amenable to computer-based analysis for classifying land cover types and assessing trends. There are several types of digital remotely sensed data (Jensen 2000). Optical remote sensing provides digital images of the amount of electromagnetic energy reflected or emitted from the Earth's surface at various wavelengths. Active remote sensing of long-wavelengths microwaves (RADAR), short-wavelength laser light (LIDAR), or sound waves (SONAR) measures the amount of backscatter from electromagnetic energy emitted from the sensor itself.

The spatial resolution (area of ground observed in a picture element or pixel), temporal resolution (how often the sensor records imagery from a particular area), spectral resolution (number of specific wavelength intervals in the electromagnetic spectrum to which the sensor is sensitive), and radiometric resolution (precision in the detected signal) determine the utility of the data for a specific application. For example, data with very high spatial resolution can be used to map habitats over local areas but low temporal resolution limits the ability to map changes over time.

A key element in the interpretation of remote sensing data is calibration and validation with *in situ* data. Ground-based data aids the interpretation of satellite data by identifying locations of specific features in the land surface. These locations can then be located on the satellite image to obtain the spectral signatures of different features. Ground-based data are also critical to test the accuracy and reliability of the interpretation of satellite data. Linking ground-based with satellite data poses logistical challenges if the locations

required are inaccessible. Moreover, the land surface is often heterogeneous so that a single pixel observed by the satellite contains multiple vegetation types. The ground observations then need to be scaled to the spatial resolution of the sensor. Despite these challenges,
ground-based data for calibration and validation are central to the effective use of satellite data for ecosystem assessment.

Analysis of satellite data are a major contribution to assessments of ecosystem conditions and trends, especially over large areas where it is not feasible to perform ground surveys. Technological challenges such as sensor drift and sensor degradation over time, lack of data continuity, and persistent cloud cover particularly in humid tropics are challenges to routine application of satellite data to monitor ecosystem condition. Accuracy and reliability of the interpretation of satellite data based on ground observations and local expertise are key to successful use for assessing ecosystem condition.

Satellite data contribute to several types of information needs for assessments of ecosystem condition, including land cover and land cover change mapping, habitat mapping for biodiversity, wetland mapping, land degradation assessments, and measurements of land surface attributes as input to ecosystem models.

Mapping of land cover and land cover change. Over the last few decades, satellite data has increasingly been used to map land cover at regional, continental, and global scales. During the 1980s, pioneering research was conducted to map vegetation at continental scales, primarily with data acquired by the U.S. National Oceanographic and Atmospheric Administration's (NOAA) meteorological satellite, the Advanced Very High Resolution Radiometer (AVHRR). Multitemporal data describing seasonal variations in photosynthetic activity were used to map vegetation types in Africa (Tucker 1985) and South America (Townshend 1987). In the 1990s, AVHRR data were used to map land cover globally at increasingly higher spatial resolution, with the first global land cover classification at 1x1 degree resolution (approximately 110x110km) (DeFries and Townshend 1994), followed by 8x8km resolution (DeFries 1998) and finally 1x1km resolution (Loveland and Belward 1997; Hansen 2000). Global satellite data also have enabled mapping of fractional tree cover to further characterize the distributions of forests over the Earth's surface (DeFries 2000). At continental and subcontinental scales, AVHRR data have been used to map the distribution of humid forests (Malingreau 1995; Mayaux 1998) and radar data provide useful information for mapping land cover types where frequent cloud cover presents difficulties for optical data (DeGrandi 2000; Saatchi 2000). A suite of recently-launched sensors, including MODIS, SPOT Vegetation, and GLI (see {Table 3.2}), provide globally comprehensive data to map vegetation types with greater accuracy due to improved spectral, spatial, and radiometric resolutions of these sensors (Friedl 2002). The GLC2000 land cover map derived from SPOT Vegetation data provides the basis for the Millennium Assessment's geographic designation of ecosystems (Fritz et al. 2004) (see {Appendix}).

One of the most significant contributions to be gained from satellite data is the identification and monitoring of land cover change, an important driver of changes in ecosystem services (see DRIVERS chapter). Data acquired by Landsat and SPOT HRV have been the primary sources for identifying land cover change in particular locations. Incomplete spatial coverage, infrequent temporal coverage, and large data volumes have precluded global analysis of land cover change. With the launch of Landsat 7 in April, 1999, data are obtained every 16 days for most parts of the Earth resulting in more

comprehensive coverage than previous Landsat sensors. Time series of Landsat and SPOT imagery have been applied to identify deforestation and regrowth mainly in the humid tropics (Skole and Tucker 1993; Achard 2002). Deforestation is the most measured process of land cover change at the regional scale, although major uncertainties exist about absolute area and rates of change
(Lepers et al. submitted).

Data continuity is a key requirement for effectively identifying land cover change. With the exception of the coarse resolution AVHRR Global Area Coverage (GAC) observations over the past twenty years, continuous global coverage has not been possible ({Table 3.2}). DeFries et al. (2002) and Hansen and DeFries (in press) have applied the AVHRR time series to identify changes in forest cover over the last two decades, illustrating the feasibility of using satellite data to detect these changes on a routine basis. Continuity of observations in the future is an essential component for monitoring land cover change and identifying locations with rapid change. For long-term data sets that cover time periods longer than the lifetime of a single sensor, cross calibration for a period of overlap is necessary. Moreover, classification schemes used to interpret the satellite data need to be clearly-defined and flexible to allow comparisons over time.

Applications for biodiversity. There are two approaches for applying remote sensing to biodiversity assessments – direct observations of organisms and communities and indirect observations of environmental proxies of biodiversity (Turner et al. 2003). Direct observations of individual organisms, species assemblages, or ecological communities are possible only with hyperspatial, very high resolution (~1m) data. Such data can be applied to identify large organisms over small areas. Air-borne observations have been used for censuses of large mammal abundances spanning several decades, for example the Kenyan remote sensing. (ref Kenyan mammal census from Mohammad dissertation).

Indirect remote sensing of biodiversity relies on environmental parameters as proxies, such as discrete habitats (e.g. woodland, grassland, or seabed grasses) or primary productivity. This approach has been employed in the US GAP analysis program (Scott and Csuti 1997). Another important indirect use of remote sensing is the detection of habitat loss and fragmentation to estimate the implications for biodiversity based on species-area relationships or other model approaches (see {BIODIVERSITY chapter}).

Wetland mapping. A wide range of remotely sensed data has been used to map wetland distribution and condition (Darras et al. 1998; Finlayson et al. 1999; Phinn et al. 1999). The utility of such data is a function of spatial and spectral resolutions and careful choices need to be made when choosing such data (Lowry and Finlayson in press). The NOAA Advanced Very High Resolution Radiometer (AVHRR), for example, observes at a relatively coarse nominal spatial resolution of 1.1 km, and allows only the broad distribution of wetlands to be mapped. More detailed observations of the extent and zonation of wetlands can be obtained using finer resolution Landsat TM (30 m) and SPOT HRV (20 m) data. As with all optical sensors, the data are frequently affected by atmospheric condition, especially in tropical coastal areas where humidity is high, and the presence of water beneath the vegetation canopy cannot be observed.

Remotely sensed data from newer spaceborne hyperspectral sensors, Synthetic Aperature Radar (SAR) and laser altimeters provide more comprehensive data on wetlands. However, although useful for providing present-day baselines, the historical archive is limited, in contrast to the optical Landsat, AVHRR, and SPOT sensors which date back to 1972, 1981, and 1986 respectively.

Aerial photographs have been acquired for many years for over half a century, at fine spatial resolutions and when cloud cover is minimal. Photographs are available in a range of formats, including panchromatic black and white, near infrared black and white, true color, and color infrared. Stereo pairs of photographs can be used to assess the vertical structure of vegetation and detect, for example, changes in the extent and height of mangroves (Lucas et al. 2002).

The European Space Agency's project Treaty Enforcement Services using Earth Observation (TEASEO) has assessed the use of remote sensing for wetland inventory, assessment and monitoring using combinations of sensors in support of wetland management. The approach has been extended through the "GlobWetland" project and its Global Wetland Information Service project to provide remotely sensed products for over 50 wetlands across 21 countries in Africa, Europe, and North and Central America. The project is designed to support on-the-ground implementation of the Ramsar Convention on Wetlands.

Assessing land degradation in drylands. Interpretation of remotely-sensed data to identify land degradation in drylands is difficult because of large variations in vegetation productivity from year-to-year variations in climate. This variability makes it problematic to distinguish trends in land productivity attributable to human factors such as over-grazing or soil salinization from variations in productivity due to interannual climate variability or cyclical drought events (Reynolds and Smith 2002). Changes in land productivity is defined by the Convention to Combat Desertification as "reduction of loss, in arid, semi-arid and dry sub-humid areas, of the biological or economic productivity of rainfed cropland, irrigated cropland, or ranges, pastures, forests, and woodlands resulting from land uses or from a process or combination of processes, including processes arising from human activities and habitation patterns." Quantifying changes in productivity involves an established baseline of land productivity against which changes can be assessed. Such a baseline is often not available. Furthermore, the inherent variability in year-to-year and even decade-to-decade fluctuations complicates the definition of a baseline. The very definition of "productivity", and whether it refers to total amount of plant growth or resilience of plant growth to abnormally low rainfall, obscures the ability to measure it.

One approach to assess land productivity is through rain-use efficiency, which quantifies net primary production (in units of biomass per unit time per unit area) normalized to the rainfall for that time period (Prince et al. 1990). Rain use efficiency makes it possible to assess spatial and temporal differences in land productivity without the confounding factor of climate variability. Several models are available to estimate net primary production (see {section 3.2.2}), some using remotely-sensed vegetation indices such as the Normalized Difference Vegetation Index (NDVI, a ratio of red to infrared reflectance indicating vegetative activity) as input data for the models. Studies have examined patterns in NDVI, rain-use efficiency, climate, and land-use practices to investigate possible trends in land productivity and causal factors (e.g., (Prince et al. 1990; Tucker et al. 1991; Nicholson et al. 1998)).

The European Space Agency's project on Treaty Enforcement Services using Earth Observation (TESEO) has examined the utility of remote sensing for mapping and monitoring desertification and land degradation in support of the Convention to Combat

Desertification (TESEO 2003). Geostationary satellites such as Meteosat operationally provide basic climatological data, which are necessary to estimate rain-use efficiency and distinguish climatic from land-use drivers of land degradation. Operational meteorological satellites, most notably the Advanced Very High Resolution Radiometer, has provided the longest term continuous record for NDVI from the 1980's to the present. More recently-launched sensors such as VEGETATION on-board SPOT and MODIS on-board the Earth Observation System {Table 3.2} have been designed specifically to monitor vegetation. Applications of microwave sensors such as ERS are emerging as possible approaches to map and monitor land productivity. Microwave sensors are sensitive to the amount of living aboveground vegetation and moisture content of the upper soil profile and are appropriate for identifying changes in semi-arid and arid conditions.

Advancements in the application of remote sensing for mapping and monitoring land degradation involves not just technical issues but institutional issues as well (TESEO 2003). National capacities to use information and technology transfer currently limit the possible applications.

Measurements of land surface and marine attributes as input to ecosystem models. Satellite data, applied in conjunction with ecosystem models (see {section 3.2.3}), provide spatially comprehensive estimates of parameters such as evapotranspiration, primary productivity, fraction of solar radiation absorbed by photosynthetic activity (FPAR), leaf area index (LAI), percentage of solar radiation reflected by the surface (albedo) (Myneni 1992; Sellers 1996), ocean chlorophyll (Doney et al. 2003), and species distributions (Raxworthy et al. 2003). These parameters are related to several ecosystem services. For example a decrease in evapotranspiration from modifying a forest to an urban system alters the ability of the forest system to regulate climate. A change in primary production relates to the food available for humans and other species. The satellite-derived parameters provide an important means for linking changes in ecosystem condition with implications for their services, for example linking changes in climate regulation with changes in land and marine surface properties (see *{Chapter CLIMATE AND AIR}*).

3.2.1.2 Geographic Information Systems

To organize and analyze remote sensing and other types of information in a spatial framework, many chapters in this report rely on geographic information systems (GIS). A GIS is a computer system consisting of computer hardware and software for entering, storing, retrieving, transforming, measuring, combining, sub-setting and displaying spatial data that have been digitized and registered to a common coordinate system (Heywood 1998; Johnston 1998). GIS allows disparate data sources to be analyzed spatially. For example, human population density can be overlain with data on net primary productivity or species endemism to identify locations within ecosystems where human demand for ecosystem services may be correlated with changes in ecosystem condition. Locations of roads can be entered into a GIS along with areas of deforestation to examine possible relationships between the two variables. The combination of remote sensing, GIS and Global Positioning System (GPS) for field validation is powerful for assessing trends in ecosystem condition (Hoffer 1994; ICSU 2002a).

GIS can be used in conjunction with remote sensing to identify land cover change. A common approach is to compare recent and historical high-resolution satellite images (e.g., Landsat Thematic Mapper). For example, {Figure 3.2} illustrates the changes in forest cover between 1992 and 1997 in Mato Grosso, Brazil, part of 100 sample sites located in the humid tropical forests to estimate tropical deforestation (Achard 2002).

Figure 3.2: Landsat TM scene from 1992, 1997, and land cover change.

GIS has also been applied in wilderness mapping, also known as "mapping human impact." These exercises estimate human influence through geographic proxies such as human population density, settlements, roads, land use, and other human made features. All factors are integrated within the GIS and either summed up with equal weights (Sanderson 2002) or weighted according to perceptions of impact (Carver 2002). This exercise has been carried out at regional scales (for example (Lesslie and Maslen 1995; Aplet 2000; Fritz 2001) as well as on a global scale (for example, (UNEP 2001; Sanderson 2002). Sanderson et al. (2002) used the approach to estimate the 10% wildest areas in each biome of the world. UNEP's Global Biodiversity (GLOBIO) project uses a similar methodology and examines human influence in relation to indicators of biodiversity (UNEP 2001).

A further application of GIS and remote sensing is to test hypotheses and responses of ecosystem services to future scenarios (Cleland 1994; Wadsworth and Treweek 1999). For example, GIS is used in the sub-global assessment of Southern Africa to predict the degree of fuelwood shortages for the different districts of Northern Sofala Province, Mozambique for the year 2030. This is done by using the GIS database showing available fuelwood per district in the year 1995 and projecting availability in the year 2030 assuming that the current trend of forest degradation of 0.05 hectares per person per year will continue. This allows identification of districts where fuelwood would be most affected.

GIS is also applicable for assessing relationships between health outcomes and environmental conditions (see {chapter HEALTH }) and for mapping risks of vulnerable populations to environmental stressors (see {chapter VULNERABILITY}). The spatial displays aim to delineate the places, human groups, and ecosystems that have the highest risk associated with them. Examples include the "red data" maps depicting critical environmental situations (Mather and Sdasyuk 1991), maps of "environmentally endangered areas" (NationalGeographicSociety 1989), and locations under risk from infrastructure expansion (Laurance et al. 2001), biodiversity loss (Myers et al. 2000), natural hazards, and impacts from armed conflicts (Gleditsch et al. 2002). The analytical and display capabilities can draw attention to priority areas that require further analysis or urgent attention. Interactive internet mapping is a promising approach for risk mapping but is currently in its infancy.

Table 3.2: Satellite sensors for monitoring land cover, land surface properties, and land and marine productivity

3.2.2 Inventories of Ecosystem Components

Inventories provide data on various ecosystem components relevant to this assessment. The most common and thorough types of inventories relate to the amount and distribution of provisioning services such as timber and agricultural products. Species inventories also provide information useful for assessing biodiversity, and demographic data provides essential information on human populations living within the systems.

3.2.2.1 Natural Resource Inventories

Many countries routinely conduct inventories of their natural resources. The inventories generally assess the locations and amounts of economically important ecosystem services such as timber, agricultural products, and fisheries. The FAO periodically publishes compilations of the national-level statistics in forest resources, agricultural production, fisheries production, and water resources (see {Table 3.3}). These statistics are widely used throughout this report. They are in many cases the only source of globally comprehensive data on these ecosystem services.

Although the assessment of ecosystem conditions and trends relies heavily on data from resource inventories, there are a number of limitations. First, questions remain about varying methods and definitions used by different countries for data collection (Matthews 2001). For example, several studies based on analysis of satellite data indicate that the FAO Forest Resource Assessment overestimates the rate of deforestation in some countries (Steininger 2001; Achard 2002; DeFries 2002). For fisheries, there are no globally consistent inventories of fisheries and fishery resources. Efforts to develop them are only starting, with the implementation of the FAO *Strategy for Improving Information on Status and Trends of Capture Fisheries*, which was adopted in 2003 in response to concerns about the reliability of fishery data (FAO 2000b). Second, resource inventories are often aggregated to the national level or by subnational administrative units. This level of aggregation does not match the ecosystem boundaries used as the reporting unit for the MA. Third, data quality is highly uneven, with greater reliability in developed countries than in developing countries. In many countries, deforestation 'data' are actually projections based on a model rather than empirical observations (Kaimowitz and Angelsen 1998). Fourth, statistics on the production of an ecosystem service do not necessarily provide information about the capacity of the ecosystem to continue to provide the service. For example, fisheries catches can increase for years through 'mining' of the stocks even though the underlying biological capability of producing fish is declining, eventually resulting in a collapse. Finally, inventories for non-commodity ecosystem services, particularly the "regulating", "supporting", and "cultural" services, have not been systematically carried out.

Table 3.3: Examples of resource inventories applicable for assessing ecosystem condition and trends

3.2.2.2 Biodiversity Inventories

Inventories of the biodiversity of ecosystems are far less extensive than inventories of individual natural resources with value as commodities. Only a small fraction of biodiversity is currently monitored and assessed. This is probably because there are few perceived economic incentives to inventory biodiversity *per se*, and because biodiversity is a complex phenomenon that is difficult to quantify and measure (see *{chapter BIODI}*). Nonetheless, biodiversity inventories can be quite useful to assessments such as the MA. They can provide a general sense of the relative biodiversity importance (e.g., richness, endemism) of ecosystems; they can illuminate the impacts of different human activities and management policies on biodiversity; and, when targeted at service-providing taxa or functional groups (e.g., pollinators), they can link changes in biodiversity within these groups directly to changes in the service provided.

Biodiversity inventories are conducted at a range of spatial scales, chosen to best address the issue or question at hand. Most, however, can be usefully grouped into three distinct categories: global inventories, regional inventories, and local inventories. Because biodiversity is complex, inventories typically focus on one aspect of biodiversity at a time, such as species richness or habitat diversity. Below we provide some examples of inventories at each of these scales, and discuss their relative strengths, limitations, and utilities for the MA.

At the global scale, only a handful of biodiversity inventories exist. These typically provide species lists for relatively well-known taxa, based on relatively large spatial units. For example, the World Conservation Monitoring Centre (1992) compiled species inventories of mammals, birds, and swallowtail butterflies for all the nations in the world. The World Wildlife Fund is conducting an inventory of all vertebrates and plants in each of the world's 867 terrestrial ecoregions (defined as relatively large units of land or water containing a distinct assemblage of natural communities and species, with boundaries that approximate the original extent of natural communities prior to major land-use change.) These inventories are useful for documenting overall patterns of biodiversity on earth, in order to indicate global priorities for biodiversity conservation or areas of high-expected threat (Sisk et al. 1994; Ceballos and Brown 1995; Dinerstein 1995). Their utility for focused analyses is limited, however, by the coarse units on which they are based and their restriction to mostly vertebrate taxa (which are not often the most important to the provision of ecosystem services). In addition, the World Conservation Union (IUCN) has been producing Red Data Books and Red Lists of Threatened Species since the 1960s. Currently, the IUCN Red List is updated annually (www.redlist.org). The criteria for listing are transparent and quantitative (see {section 3.2.3.4} below). The IUCN Red List is global in coverage, and is the most comprehensive list of threatened species, with 100% of known bird and mammal species evaluated, and plans for complete coverage of amphibians and reptiles in the next few years. Data on fish species include FISHBASE (Frose and Pauly 2000), CephBASE (Wood et al. 2000), ReefBase (Oliver et al.), and the Census of Marine Life (O'Dor 2004).

Inventories at regional or continental scales are generally of higher overall quality and are more common than global data. Many of these datasets are based on grids of varying resolution. Examples include data on vertebrates in sub-Saharan Africa (grid size 1 degree or approximately 110 km^2, (Balmford et al. 2001), birds in the Americas (grid size 611,000km^2, (Blackburn and Gaston 1996), several taxa of plants and animals in Britain (grid size 10 km^2, (Prendergast et al. 1993), and terrestrial vertebrates and butterflies in Australia (grid size 1 degree) (Luck et al. in review). These grid-based inventories, as well as others based on political boundaries (e.g., countries, states) are based on arbitrary units that rarely reflect ecosystem boundaries. As a result, their utility is limited in assessing the biodiversity of a particular ecosystem, as may often be the goal in MA analyses. Some regional-scale inventories are based on ecological units, including a study on vertebrates, butterflies, tiger beetles, and plants for 116 ecoregions in North America (Ricketts et al. 1999). All of these regional inventories can be used to understand patterns of biodiversity and endangerment (e.g., (Ceballos and Brown 1995) and to link these patterns to threats and drivers operating at regional scales (e.g., (Balmford et al. 2001; Ricketts in review). As is often the case, these datasets are most complete and dependable in the developed world, although data are improving in many developing regions of central interest to the MA.

Because many ecosystem services are provided locally (e.g., pollination, water purification), local-scale biodiversity inventories are often the most directly valuable for assessing those services. There are thousands of local inventories in the literature, comparing biodiversity between ecosystem types, among land use intensities, and along various environmental gradients. This literature has not been systematically compiled, and it is not possible to list all the studies here. We illustrate the types of available data here with biodiversity studies in agricultural landscapes dominated by coffee cultivation. Local inventories in these landscapes have quantified the decline in both bird (Greenberg et al. 1997) and arthropod (Perfecto et al. 1997) diversity with increasing intensification of coffee production. Other studies have shown a decline in moth (Ricketts et al. 2001) and bird (Luck and Daily 2003) diversity with increasing distance from remnant patches of forest. Most relevant to ecosystem services, the diversity and abundance of coffee-visiting bees declines with increasing distance from forest (Ricketts in press), and with increasing intensification (Klein et al. 2002). Local inventories offer data that can directly inform land-use policies and illuminate trade-offs among ecosystem services for decision-makers. Unfortunately, they are often time and resource intensive. In addition, the results are only relevant to the specific taxon and location under study, so general lessons are difficult to glean. However, the collective results of many such studies can lead to useful general guidelines and principles.

Another method of compiling results from many biodiversity inventories is to examine the collections of museums and herberia (Ponder et al. 2001). Scientists conducting biodiversity surveys typically deposit their collections in these institutions, along with data on location, habitat, date, etc. Museums and herbaria therefore house enormous amounts of information, aggregated sometimes over centuries of study. Furthermore, museums are beginning to use information technologies and the internet to pool their information into aggregate databases, such that records from any museum can be searched, e.g. (Edwards et al. 2000). These aggregate databases are an invaluable resource for studying the distribution of biodiversity. Museum and herbaria records, however, often contain a variety of spatial, temporal, and taxonomic biases and gaps, due to the ad hoc nature of collecting and the interests of collecting scientists (Ponder et al. 2001). These biases must be carefully considered when using museum data to assess status and trends of biodiversity.

The chapters in this report rely on available data sources for characterizing biodiversity in the individual systems and its response to changes in ecosystem condition. Ideally, these data would be collected routinely according to an appropriate sampling strategy that meets the needs of the specified measures. However, most often this is not the case, and data assimilated for other purposes are used, such as routine or sporadic surveys and observations made by naturalists. Generally such observations relate only to the most obvious and common species, especially birds, sometimes mammals, butterflies etc.

3.2.2.3 Demographic and Socioeconomic Data on Human Populations

Because this assessment considers human populations as integral components of ecosystems, data on the populations living within the systems are one of the foundations for this analysis. Demographic and socioeconomic data provide information on the distributions of human populations within ecosystems, a prerequisite to analyzing the dependence of human well-being on ecosystem services. Most information on the distribution and characteristics of human population is collected through population censuses and surveys. Nearly all countries of the world conduct periodic censuses (see http://www.census.gov/ipc/www/cendates/cenall.pdf); most countries conduct them once per decade. Census data are collected and reported by administrative or political units, such as counties, provinces or states. These administrative boundaries generally do not correspond to the geographic boundaries of ecosystems. To address this mismatch, the most recent version of the Gridded Population of the World (GPW, version 3) (CIESIN et al. 2004; CIESIN_and_CIAT 2004) contains population estimates for over 350,000 administrative units converted to a grid of latitude-longitude quadrilateral cells at a nominal spatial resolution of 5 km^2 at the equator (Deichmann et al. 2001). The accuracy depends on the quality and year of the input census data and the resolution of the administrative units. Other datasets allocate population toward urban areas, roads, and other likely population centers, such as LandScan that uses many types of ancillary data, including land cover, roads, night-time lights, elevation and slope, to reallocate populations within administrative areas to more specific locations (Dobson 2000).

There are large data gaps on poverty distribution and access to ecosystem services such as water (UNDP 2003). Some census data include resource use such as fuelwood and water source (Govt._of_India 2001), but inventories on the use of ecosystems services are not generally available to establish trends. Increasingly, however, censuses and large-scale surveys are beginning to include questions on resource use. The World Bank's Living Standards Measurement Survey (LSMS), for example, is introducing modules on resource use (Grosh and Glewwe 1995). As most nationally representative socioeconomic and demographic surveys are not georeferenced beyond administrative units, they must be used with care when making inferences at the moderate and high resolutions often used in ecological data analysis.

By combining census information about human settlements with geographic information, such as stable-city night time lights from satellite data, a new global database indicates urban areas from rural ones (CIESIN et al. 2004). These can be applied to distinguish urban and rural land areas in different ecosystems, and infer implications for resource use (see {URBAN CHAPTER}).

3.2.3 Numerical Simulation Models

Numerical models are mathematical expressions of processes operating in the real world. The ecological and human interactions within and among ecosystems are complex, and they involve physical, biological and socio-economic processes occurring over a range of temporal and spatial scales. Models are designed as simplified representations to examine assumptions and responses to driving forces.

Models span a wide range in complexity with response to processes and spatial and temporal scales. Simple correlative models use statistical associations established where data are adequate to predict responses where data are lacking. For example the CLIMEX model (Sutherst 1995) predicts the performance of an insect species in a given location and year in response to climate change based on previously-established correlations from comparable locations and previous years. Dynamic, process-based models, on the other hand, are sets of mathematical expressions describing the interactions among components of a system at a specified time step. For example, the CENTURY model simulates fluxes of carbon, water, and nitrogen among plant and soil pools within a grassland ecosystem (Parton 1988). An emerging class of models incorporate the dynamic processes, but also simulate the dynamics of interacting species or plant functional types, such as IBIS (Foley 1996) and LPJ (Sitch et al. 2003). Such models have been applied at the site, regional, and global scales to investigate ecosystem responses to climate change scenarios and increasing atmospheric carbon dioxide concentrations (e.g. (Cramer et al. 2004)).

Categories of models useful for the assessment of ecosystem condition and services are given in {Table 3.4}. These models address various aspects of ecosystem condition. For example, hydrologic models can be used to investigate the effects of land cover changes on flood protection, population models can assess the effects of habitat loss on biodiversity, and integrated assessment models can synthesize this information for assessing effects of policy alternatives on ecosystem condition. ({Table 3.4}). The assessment relies on models to:

- **Fill data gaps.** As noted above, data to assess trends in ecosystem condition and their services are often inadequate, particularly for regulating, supporting, and cultural services. Models are used to address these deficiencies. For example, chapter 14 in this volume, on climate regulation uses results from four ecosystem models (McGuire 2001) to estimate the impacts of changes in land use, climate, and atmospheric composition on carbon dioxide emissions from ecosystems.

- **Quantify responses of ecosystem services to management decisions.** One of the major tasks for the MA is to assess how changes in ecosystem condition alter services. Does removal of forest cover within a watershed alter flood protection? Does conversion to cropland alter climate regulation? Models can be used to simulate changes in the ecosystem condition (e.g., land cover) and estimate the response (e.g., stream flow). A hydrologic model (e.g., (Liang 1996) can quantify the stream flow in response to removal of forest cover. A land surface model linked to a climate model (e.g., (Sellers 1986) can quantify the change in water and energy fluxes to the atmosphere from a specified change in land cover and the resulting effect on surface temperature. To the extent that models are adequate representations of reality, they provide an important tool for quantifying the effects of alternative management decisions on ecosystem services.

- **Predict long-term ecological consequences of altered ecosystem condition.** Many human activities impact ecosystem condition only after a time lag. As a consequence, effects of ecosystem management are not observed for many years. In such cases, models can be used to predict the long-term ecological consequences of human impact on ecosystems. For example, the effect of timber harvest on the persistence of threatened species such as the spotted owl can be assessed using habitat-based metapopulation models (Akçakaya and Raphael 1998). The reliability of long-term model predictions depends on the level of understanding of the system, the amount and quality of available data, the time horizon, and incorporation of uncertainty. Predictions about simpler systems (e.g., single-species dynamics) are more reliable than those about complex systems (such as community composition and dynamics), because of the higher level of understanding ecologists have for the simpler systems. The amount and quality of the data determine the uncertainty in input parameters, which in turn affect the reliability of the outcome. Longer-term predictions are less reliable because these uncertainties are compounded over time. However, even uncertain predictions can be useful, if the level of uncertainty can be objectively quantified. Complex models can also identify shifts in ecosystem regime, such as the sudden loss of submerged vegetation in shallow lakes subject to eutrophication (Scheffer et al. 2001), and non-linear responses to drivers.

- **Test sensitivities of ecosystem condition to individual drivers or future scenarios.** Observed changes in ecosystem condition result from the combined responses to multiple drivers. Changes in soil fertility in a rangeland, for example, reflect the combined response to grazing pressure, climate variations, and changes in plant species. Direct observations of soil fertility do not enable understanding of which driver is causing the response or how the drivers interact. A series of model simulations, changing one or more drivers for each model run, facilitates understanding of the response of soil fertility to each of the drivers. To the extent that models represent processes realistically, model simulations can identify non-linear and threshold responses of ecosystems to multiple drivers. For example, either overfishing or pollution alone may not lead to precipitous declines in fish stocks, but the combined response could have unanticipated effects on fish stocks.

- **Assess future viability of species**. When a species has particular importance (e.g., as an indicator, sensitive, endemic, threatened, or economically important species), the change in future prospects of the species may be of interest. Quantitative methods and models for assessing the chances of persistence of species in the future are collectively called Population Viability Analysis (PVA). Models used in PVAs range from unstructured single-population models to metapopulation models with explicit spatial structure based on the distribution of suitable habitat (Boyce 1992; Burgman 1993). PVA provides a rigorous methodology that can use different types of data, incorporate uncertainties and natural variabilities, and make predictions that are relevant to conservation goals. PVA is most useful when its level of detail is consistent with the available data, and when it focuses on relative (i.e., comparative) rather than absolute results, and on risks of decline rather than extinction (Akçakaya and Sjogren-Gulve 2000). An important advantage of PVA is its rigor. In a comprehensive validation study, Brook et al. (2000) found the risk of population decline predicted by PVA closely matched observed outcomes, there was no significant bias, and population size projections did not differ significantly from reality. Further, the predictions of five PVA software packages they tested were highly concordant. PVA results can also be tested for single models by comparing predicted values with those observed or measured in the field (McCarthy 2001).

- **Understanding the dynamics of social environmental interactions.** Individual based methods such as multi-agent modeling are increasingly used to understand social and environmental interactions. Multi-agent behavioral systems (MABS) seek to model social-environment interactions as dynamic processes (see Moss et al. 2001). Human actors are represented as software agents with rules for their own behaviour, interactions with other social agents, and responses to the environment. Physical processes (such as soil erosion) and institutions or organizations (such as an environmental regulator) may also be represented as agents. A multi-agent system could represent multiple scales of vulnerability and produce indicators of multiple dimensions of vulnerability for different populations. Multi-agent behavioral systems have an intuitive appeal in participatory integrated assessment. Stakeholders may identify with "their" agents and be able to validate a model in qualitative ways that is difficult to do for econometric or complex dynamic simulation models. However, such systems require significant computational resources (proportional to the number of agents) and a paucity of data for validation of individual behaviour is a constraint.

Table 3.4: Examples of numerical models for assessing conditions and trends in ecosystems and their services

Models are useful tools for ecosystem assessments, if the selection of models, input data, and validation are considered carefully for particular applications. A model developed with data from one location is not directly applicable to other locations. Moreover, data to calibrate and validate models are often difficult to obtain. The appropriateness of a model for an assessment task also depends as much on the capacity of the model variables to capture the values and interests of the decision-making and stake-holding communities as on the accuracy of the underlying scientific data.

3.2.4 Indicators of Ecosystem Condition and Services

We define an indicator as a scientific construct that uses quantitative data to measure ecosystem condition and services, drivers of changes, and human well-being. Properly constituted, an indicator can convey relevant information to policymakers. In this assessment, indicators serve many purposes, for example::

- as easily-measured quantities to serve as surrogates for more difficult to measure characteristics of ecosystem condition. For example, the presence of fecal coliform in a stream is relatively easy to measure and serves a surrogate for poor sanitation in the watershed, which is more difficult to measure.

- as a means to incorporate several measured quantities into a single attribute as an indicator of overall condition. For example, the widely-used Index of Biotic Integrity (IBI) is an indicator of aquatic ecosystem condition (Karr et al. 1986). The IBI is an additive index combining measures of abundances of different taxa. The individual measures can be weighted according to the importance of each taxa for aquatic health.

- as a means to effectively communicate with policy makers regarding trends in ecosystem conditions and services. For example, information on trends in disease incidence reflects trends in disease control as a “regulating” ecosystem service. The former can be readily communicated to a policymaker.

- as a means to measure the effectiveness of policy implementation.

Identifying and quantifying the appropriate indicators is one of the most important aspects of the chapters in this report because it is simply not possible to measure and report all aspects of ecosystems and their relation to human well-being. It is also important to identify appropriate indicators to establish a baseline against which future ecosystem assessments can be compared.

Indicators are designed to communicate information quickly and easily to policy makers. Economic indicators, such as GDP, are highly influential and immediately understood by decision makers. Measures of poverty, life expectancy, and infant mortality directly convey information about human well-being. Some environmental indicators, such as global mean temperature and atmospheric carbon dioxide concentrations, are becoming widely accepted as measures of anthropogenic effects on global climate. Measures of ecosystem condition are far less mature, although some biophysical measures such as spatial extent of an ecosystem and agricultural output are relatively easy to quantify. There are at this time no widely accepted indicators to measure trends in supporting, regulating, or cultural ecosystem services, much less indicators that measure the effect of changes in these services on human well-being. Effective indicators meet a number of criteria (NRC 2000) (see {Box 3.1}).

Box 3.1: Criteria for effective ecological indicators

Major indicators used throughout the report for assessing ecosystem conditions, their service, and quantifying responses to drivers ({Table 3.5}) include indicators of direct drivers of change, or ecosystem condition, and of ecosystem services. ({Section 3.x} of this chapter discusses indicators of human well-being and their utility for measuring how well-being responds to changes in ecosystem services).

*Indicators of direct drivers of change***.** No single indicator represents the totality of the various drivers. Some direct drivers of change (see (Millennium Ecosystem Assessment 2003) and chapter on DRIVERS) have relatively straightforward indicators, such as fertilizer usage, water consumption, irrigation, and harvests. Indicators for other drivers, including invasion by non-native species, climate change, land cover conversion, and landscape fragmentation are not as well-developed and data to measure them are not as readily available.

Indicators of ecosystem condition. Indicators of biophysical condition of ecosystems do not directly reflect the cause-and-effect of the drivers but nevertheless contribute to policy formulation. Such indicators are not designed to represent cause-and-effect relationships between drivers of changes and their consequences. Rather, they serve to direct attention to changes of importance. To determine causal relationships, models of interactions among variables must be used. As an analogy with human health, an increase in body temperature indicates infection that warrants further examination. As an example in the biophysical realm, declining trends in fish stocks can trigger investigations of possible causal mechanisms and policy alternatives. Indicators of ecosystem condition include many dimensions, ranging from the extent of the ecosystem to demographic characteristics of human populations to amounts of chemical contaminants (The H. John Heinz III Center for Science 2002).

Indicators of ecosystem services: Indicators for the "provisioning" services discussed in Part II generally relate to commodity outputs from the system (e.g., crop yields, fisheries) and are readily communicable to policymakers. Indicators related to the underlying biological capability of the system to maintain the production, the "supporting" and "regulating" services, are a greater challenge. For example, indicators measuring the capability of a system to regulate climate, such as evapotranspiration or albedo, are not as readily interpretable for a policymaker.

Table 3.5: Examples of indicators used for assessing ecosystem condition and trends

Though indicators are essential, they need to be used with caution (Bossel 1999). Over-reliance on indicators can mask important changes in ecosystem condition. Second, while it is important that indicators are based on measurable quantities, the selection of indicators can be biased towards attributes that are easily quantifiable rather than truly reflective of ecosystem condition. Third, comparing indicators and indices from different temporal and spatial scales is challenging because units of measurement are often inconsistent. Adding up and combining factors has to be done very carefully and it is crucial that the method for combining individual indicators is well understood.

Indicators of biodiversity. Indicators of biodiversity are particularly important for this assessment (see {section 3.3.4} for indicators of human well-being). Indicators of the amount and variability of species within a defined area can take many forms. The most common measures are:

- Species richness: the number of species
- Species diversity: the number of species weighted by their relative abundance, biomass, or other characteristic, e.g., Shannon-Weiner or other similar indices (Rosenzweig 1995).

These simple measures do not capture many aspects of biodiversity. They do not differentiate between native and invasive/introduced species, differentiate among species in terms of sensitivity or resilence to change, or focus on species that fulfill significant roles in the ecosystem (e.g., pollinators, decomposers). Moreover, the result depends on the definition of the area and may be scale-dependent. The measures also may not always reflect biodiversity trends accurately. For example, ecosystem degradation by human activities may temporarily increase species richness in the limited area of the impact. Thus, refinements of these simple measures provide more insights into the amount of biodiversity (see {Box 3.2}).

Aggregate indicators of trends in species populations such as the Index of Biotic Integrity for aquatic systems (Karr and Dudley 1981) and the Living Planet Index (Loh 2002) use existing data sets to identify overall trends in species abundance and, by implication, the condition of the ecosystems in which they occur. The Living Planet Index is an aggregation of three separate indices, each the average of trends in species abundances in forest, freshwater, and marine biomes. The index can be applied at national, regional, and global levels. The effectiveness of such an aggregate indicator depends on availability and access to data sets on a representative number of species, particularly problematic in many developing countries.

Box 3.2: Examples of indicators of biodiversity

The number of species threatened with extinction is an important indicator of biodiversity trends. However, using this indicator requires a number of conditions to be met. First, the criteria used to categorize species into threat classes must be objective, transparent, and have a scientific basis. Second, the changes in the status of species must reflect genuine changes in the conservation status of the species (rather than changes in knowledge or taxonomy, for example). Third, the pool of species evaluated in two different time periods must be comparable (if more threatened species are evaluated first, the proportion of threatened species may show a spurious decline). The IUCN Red List of Threatened Species mentioned above meets these conditions. The criteria used in assigning species to threat categories (IUCN 2001) is quantitative and transparent, yet allows for flexibility and can incorporate data uncertainties (Akçakaya 2000). The IUCN Red List database also records whether or not a species has been evaluated for the first time. For species evaluated previously, the assessment includes reasons for any change in status, such as (a) genuine change in the status of the species; (b) new or better information available; (c) incorrect information used previously; (d) taxonomic change affecting the species; and (e) previous incorrect application of the Red List criteria. Finally, the complete coverage of some taxonomic groups helps make evaluations comparable, although the fact that new species are being evaluated for other groups must be considered when calculating measures such as the proportion of threatened species in those groups.

3.2.5 Indigenous, Traditional, and Local Knowledge

Traditional Knowledge (TK) broadly represents information from a variety of sources including indigenous peoples, local residents, and traditions. The term indigenous knowledge (IK) is also widely used referring to knowledge owned by ethnic minorities from the approximately 300 million indigenous people worldwide (Emery, 2000).

Traditional knowledge (TK), Indigenous Knowledge (IK) and Local Knowledge is receiving increased interest as a valuable source of information (Martello 2001) about ecosystem condition, sustainable resource management (Johannes 1998; Berkes 1999; 2002), soil classification (Sandor and Furbee 1996), land use investigations (Zurayk et al. 2001) and the protection of biodiversity (Gadgil et al. 1993). The International Council for Science (ICSU 2002b) defines traditional knowledge as:

a cumulative body of knowledge, know-how, practices and representation maintained and developed by peoples with extended histories of interaction with the natural environment. These sophisticated sets of understandings, interpretations and meanings are part and parcel of a cultural complex that encompasses language, naming and classification systems, resource use practices, ritual, spirituality and worldview.

Pharmaceutical companies and agribusiness and environmental biologists have all found traditional knowledge to be a rich source of information (Cox 2000; Kimmerer 2000). Traditional knowledge provides empirical insight into crop domestication, breeding and management. It is particularly important in the field of conservation biology for developing conservation strategies appropriate to local conditions. Traditional knowledge is also useful for assessing trends in ecosystem condition (Mauro and Hardinson 2000) and for restoration design (Kimmerer 2000) since it tends to have qualitative information of a single local record over a long time period.

Oral histories can play a particularly important role in the field of vulnerability assessment, as they are especially effective at gathering information on local vulnerabilities over past decades. Qualitative information derived from oral histories can be further developed as storylines for further trends and can lead into role playing simulations of new vulnerabilities or adaptations
(Downing et al., 2001-MISSING REF SF)

However, TK has for a long time not been treated equally to knowledge derived from formal science. Article 27 of the Universal Declaration of Human Rights (UDHR) of 1948 protects Intellectual Property (IP) (Steven and Justine 2003-MISSING REF SF). Nevertheless intellectual property rights of indigenous people have been violated in the past (Cox 2000). The Convention on Biological Diversity of 1992 for the first time established international protocols that allow the protection and sharing of national biological resources and specifically addresses issues of traditional knowledge. In particular, parties to the convention agree to respect and preserve traditional knowledge and to promote wide applications and equitable sharing of benefits from traditional knowledge (Antweiler 1998; Cox 2000; Singhal 2000).

The integration of Traditional Environmental Knowledge (TEK) with formal science can provide a number of benefits particularly in the field of sustainable resource management (Johannes 1998; Berkes 2002). However, integrating TEK with formal science is not easy and is sometimes problematic (Antweiler 1998; Fabricus et al. 2004). Moreover, existing practices of TEK such as forest management are not necessarily sustainable (Antweiler 1998). Johnson (1998) gives the following reasons why the integration of TEK is difficult: (1) TEK is disappearing and there is a lack of resources to document it before it is lost; (2) it is not easy to translate ideas and concepts from a culture based on TEK (mainly oral based knowledge systems) into the concepts and ideas of formal science; (3) even between social and natural scientists there is disagreement regarding appropriate methods to document and integrate TEK as natural scientists often criticize the lack of rigor of the traditional anthropological methods for interviewing and participant observation; and (4) the integration of TEK and formal science is linked to the question of political power and TEK is often seen as subordinate to formulate science.

It has been repeatedly pointed out that if TEK is integrated it needs to be understood within its historical, socioeconomic, political, and environmental and cultural location (Berkes 2002). This implies that the ratio of local to scientific knowledge will vary depending on the case and situation (Antweiler 1998). It is also important that the limitations and shortcomings of integrating TEK and formal science are first addressed and the methods chosen to collect this knowledge should take the location specific environments in which they operate into account (Singhal 2000). Integration can also be hindered by different representations of cross-scale interactions, non-linear feedbacks, and uncertainty in TEK and formal science (Gunderson and Holling 2002). Due to this high degree of uncertainty it is essential to validate and compare both formal and informal knowledge (Fabricus et al. 2004).

There have been general concerns about scaling TEK up to broader spatial scales, since TEK is seldom relevant outside of the local context (Forsyth 1999; Lovell et al. 2002). Moreover, analysts warn of a downplaying of environmental problems when TEK is over-emphasized, and are concerned about politicians using flawed TEK as a reason for ignoring environmental challenges. On the other hand researchers have also warned that the efforts to integrate or bridge different knowledge systems will lead inevitably to the compartmentalization and distillation of traditional knowledge into a form that is understandable and usable by scientists and resource managers alone (Nadasdy 1999).

The utility of TEK and local knowledge is hampered when the change in the social-ecological system is faster than the rate of knowledge evolution. For example when sacred pools in the Kat River Valley in South Africa became surrounded by the invasive Australian *Acacia mearnsi*i, this species was afforded the same local protection as the valuable indigenous species that are naturally associated with such pools (Fabricus et al. 2004). Furthermore local knowledge rarely responds to slow processes such as soil erosion, invasive plants, siltation of water bodies, encroachment of mines on rangeland and slow changes on ground water quality due to cattle dip.

Despite these limitations, TEK, if interpreted carefully and assessed rigorously, can provide important data on ecosystem conditions and trends. The most promising methods of data collection are participatory approaches, in particular Participatory Rural Appraisal (PRA) (Catley 1996). PRA is an alternative to unstructured visits to communities, which may be biased towards more accessible areas, or costly and time-consuming questionnaire surveys (Chambers 1994). PRA was developed during the early 1990s from Rapid Rural Appraisal (RRA), a cost-effective and rapid way of gathering information. RRA was criticised as being too "quick and dirty" and not sufficiently involving the local people. PRA is similar to RRA, but tries to overcome the criticism towards RRA by allowing the recipient more control of the problem definition and solution design and by carrying out the research over a longer period (Zarafshani 2002). Activities such as interviewing, transects, mapping, measuring, analysis, and planning are done jointly with the local people (Cornwall and Pratt 2003). PRA is based on an action research approach in which theory and practice are constantly challenged through experience, reflection and learning (Scoones 1995). However, these methods are in their purest sense never bottom-up. Even the most recent form, called Participatory Action Research (PAR) that places more emphasis on the subject, does not overcome this drawback entirely (Pain and Francis 2003).

The participatory methods also have their limitations: First, they only produce certain types of information which can be brief and superficial. Second, the information collected may reflect peoples' own priorities and interests. Third, there might be an unequal power representation amongst participants and between participants and researchers (Cooke and Kothari 2001). Glenn (2003) warns that a rush to obtain traditional knowledge can be biased towards pre-existing stereotypes and attention to vocal individuals who do not necessarily reflect consensus.

In the Millennium Assessment sub-global assessments, a wide range of participatory research techniques were used to collect and integrate TEK and local knowledge into the assessment process. Besides the general techniques of PRA (Pereira 2004), techniques such as focus group workshops (Borrini-Feyerabend 1997), semi-structured interviews with key informants (Pretty 1995) and forum theatre, free hand and GIS mapping, pie charts, trendlines, timelines, ranking, Venn diagrams, problem trees, pyramids, role-playing and seasonal calendars are used (Borrini-Feyerabend 1997; Jordan and Shrestha 1998; Motteux 2001).

3.2.6 Case Studies of Ecosystem Responses to Drivers

Case studies provide in depth analyses of responses of ecosystem conditions and services to drivers in particular locations. For example, the study of Yaqui Valley in Mexico illustrates the response of birds, marine mammals, and fisheries to upland runoff generated by increasing fertilizer use in the heavily irrigated valley (Turner II et al. 2003). Evidence generated from a sufficient number of case studies allows general principles to emerge about ecosystem responses to drivers. Case studies, which can analyze relationships in more detail than would be possible with nationally-aggregated statistics or coarse resolution data, also illustrate the range of ecosystem responses to drivers in different locations or under different biophysical conditions.

Few studies have been undertaken to synthesize information from case studies. One such effort analyzed 152 subnational case studies investigating the response of tropical deforestation to economic, institutional, technological, cultural, and demographic drivers

(Geist and Lambin 2001; 2002). The analysis revealed complex relationships between drivers and deforestation in different regions of the tropics, indicating challenges for generic and widely applicable land-use policies to control deforestation. The MA does not carry out such extensive meta-analyses, but rather uses their results where available as well as results from individual case studies from the scientific literature.

Drawing conclusions from case studies must be done with caution. First, individual studies do not generally use standard protocols for data collection and analysis so that comparisons across case studies are difficult. Second, researchers make decisions about where to carry out a case study on an individual basis so that biases might be introduced from inadequate representation from different locations. Third, unless a sufficient number of case studies are available it is not prudent to draw general conclusions and extrapolate results from one location to another. In spite of these limitations, the MA relies on published case studies to illustrate possible linkages between ecosystem response and drivers and to fill gaps generated by lack of more comprehensive data when necessary.

3.3 Assessing the Value of Ecosystem Services for Human Well-being

This section addresses the data and methods for assessing the linkages between ecosystem services and human well-being ({Figure 3.1}).

3.3.1 Linking Ecosystem Condition and Trends to Well-being

Ecosystem condition is only one of many factors that affect human well-being, making it challenging to assess linkages between them. Health outcomes, for example, are the combined result of ecosystem condition, access to health care, economic status, and myriad other factors. Interpretations of trends in indicators of well-being must appropriately account for the full range of factors involved.

The impacts of ecosystem change on well-being are often subtle, which is not to say unimportant; impacts need not be drastic to be significant. A small increase in food prices resulting from lower yields as a result of land degradation will affect the well-being of many people, even if none starve as a result.

Tracing the linkages between ecosystem conditions and trends and human well-being is often difficult. Two basic approaches are used. The first attempts to correlate trends in ecosystem condition to changes in human well-being directly, while the second attempts to trace the impact through biophysical and socioeconomic processes to the groups affected. For example, the impact of water contamination on the incidence of human disease could be estimated by correlating measures of contaminants in water supplies with measures of the incidence of gastrointestinal illnesses in the general population. Alternatively, the impact could be estimated by using a dose-response function that relates the incidence of illness to the concentration of contaminants to estimate the increase in the probability of illness, then combining that with estimates of the population served by the contaminated water to arrive at a predicted total number of illnesses.

Both approaches face considerable problems. Efforts to correlate ecosystem condition with human well-being directly are difficult because of the presence of multiple confounding factors. Thus the incidence of illness depends not only on the concentration of airborne contaminants but also on predisposition to illness through factors such as nutritional status or the prevalence of smoking, exposure factors such as the proportion of time spent outdoors, and so on. Analyses linking well-being and ecosystem condition are most easily carried out at a local scale, where the linkages can be most clearly identified.

3.3.2 Measuring Well-being

Human well-being has several key components: the basic material needs for a good life, freedom and choice, health, good social relations, and personal security. Well-being exists on a continuum with poverty, which has been defined as "pronounced deprivation in well-being." One of the key objectives of the MA is to identify the direct and indirect pathways by which ecosystem change can affect human well-being, whether positively or negatively.

Well-being is multidimensional, and so very hard to measure. All available measures have problems, both conceptual (are they measuring the right thing, in the right way?) and practical (how do we actually implement them?). Moreover, most available measures are extremely difficult to relate to ecosystem services.

Economic valuation offers a way both to value a wide range of individual impacts (some quite accurately and reliably, others less so) and, potentially but controversially, to assess well-being as a whole by expressing the various 'apples and oranges' that make up well-being in a single unit (typically a monetary unit). It has the advantage that impacts denominated in monetary units are readily intelligible and comparable to other benefits or to the costs of intervention. It can also be used to provide information to examine distributional, equity, and intergenerational aspects. {Section 3.3.3} discusses economic valuation techniques.

Health indicators address an important subset of impacts of ecosystem services on well-being. They are an important complement to economic valuation because they concern impacts that are very difficult and controversial to value. Some health indicators address specific types of health impacts, others attempt to aggregate a number of health impacts. Likewise, poverty indicators measure a dimension of well-being that is often of particular interest. {Section 3.3.4} discusses health, poverty, and other indicators.

Numerous other well-being indicators (e.g., the Human Development Indicator) have been developed, in an effort to capture the multidimensionality of well-being into a single number, with varying degrees of success. Although these indicators are arguably better measures of well-being, they tend not to be very useful for assessing the impact of ecosystems, as many of the dimensions they add (e.g., literacy) tend not to be sensitive to ecosystem condition. {Section 3.3.5} examines these aggregate indicators and the limitations they face.

3.3.3 Economic Valuation

One of the main reasons we worry about the loss of ecosystems is that they provide valuable services – services that may be lost as ecosystems degrade. The question then immediately arises: how valuable are these services? Or, put another way, how much worse off would we be if we had less of these services? We need to be able to answer these questions to inform the choices we make in how to manage ecosystems.

Economic valuation attempts to answer these questions. It is based on the fact that human beings derive benefit (or "utility") from the use of ecosystem services either directly or indirectly, whether currently or in the future, and that they are willing to 'trade' or exchange something for maintaining these services. As utility cannot be measured directly, economic valuation techniques are based on observation of (market and non-market) exchange processes. Economic valuation usually attempts to measure all services in monetary terms, in order to provide a common metric in which to express the benefits of the very diverse variety of services provided by ecosystems. This explicitly does not mean that only services that generate monetary benefits are taken into consideration in the valuation process. On the contrary, the essence of practically all work on valuation of environmental and natural resources has been to find ways to measure benefits which do not enter markets and so have no directly observable monetary benefits.

Economic valuation has also been used to derive the total value of ecosystem services (Costanza et al. 1997) and to simulate the value of ecosystem services in an integrated earth system model (Boumans et al. 2002). In this chapter, we focus on methods useful for assessing the value of *changes* in ecosystem services resulting from management decisions or other human actions, as opposed to the absolute value of ecosystem services.

3.3.3.1 Valuation Methods

Many methods for measuring the utilitarian values of ecosystem services are found in the resource and environmental economics literature (Hufschmidt et al. 1983; Braden and Kolstad 1991; Hanemann 1992; Dixon et al. 1994). {Table 3.6} summarizes the main economic valuation techniques. Some are broadly applicable, some are applicable to specific issues, and some are tailored to particular data sources. As in the case of private market goods, a common feature of all methods of economic valuation of ecosystem services is that they are founded in the theoretical axioms and principles of welfare economics. These measures of change in well-being are reflected in people's willingness to pay (WTP) or willingness to accept (WTA) compensation for changes in their level of use of a particular service or bundle of services (Hanemann 1991; Shogren and Hayes 1997).

Table 3.6: Principal economic valuation techniques.

A number of factors and conditions determine the choice of specific measurement methods. For instance, when the ecosystem service in question is privately owned and traded in the market, its users have the opportunity to reveal their preferences for that service compared to other substitutes or complementary commodities through their actual market choices, given relative prices and other economic factors. For this group of ecosystem services a demand curve can be derived from observed market behavior, and this allows changes in well-being to be estimated. However, many ecosystem services are not privately owned and not traded and hence their demand curves cannot be directly observed and measured. Alternative methods have been used to derive values for such ecosystem services.

Figure 3.3: Valuing the impact of ecosystem change.

Some of the available valuation measures are based on actual observed behavior data, including some methods that deduce values indirectly from behavior in surrogate markets, which are hypothesized to have a direct relationship with the ecosystem service of interest. Others are based on hypothetical rather than actual behavior data, where people's responses to questions describing hypothetical markets or situations are used to infer value. These are generally known as 'stated preference' techniques, in contrast to those based on behaviour, which are known as 'revealed preference' techniques.

Valuation is a two-step process. First, the services being valued have to be identified. This includes understanding the nature of the services and their magnitude, and how they would change if the ecosystem changed; who makes use of the services, in what way and for what purpose, and what alternatives they have; and what tradeoffs might exist between different kinds of services an ecosystem might provide. The bulk of the work involved in valuation actually concerns quantifying the biophysical relationships. In many cases, this requires tracing through and quantifying a chain of causality such as that shown in {Figure 3.3}. Valuation in the narrow sense only enters in the second step in the process, in which the value of the impacts is estimated in monetary terms.

Changes in productivity. The most widely used technique, thanks to its broad applicability and its flexibility in using a variety of data sources, is known as the *change in productivity* technique. It consists of tracing through chains of causality such as those illustrated in {Figure 3.3}, so that the impact of changes in the condition of an ecosystem can be related to various measures of human well-being. Such impacts are often reflected in goods or services that contribute directly to human well-being, and as such are often relatively easily valued. The valuation step itself depends on the type of impact, but is often straightforward:

- The net value in reductions in irrigated crop production resulting from reduced water availability is easy to estimate, for example, as crops are often sold. (Even so, it is a very common error to over-estimate this impact by using the reduction in the gross rather than the net value of crop production.)

- Where the impact is on a good or service that is not marketed, or where observed prices are unreliable indicators of value, the valuation can become more complex. The impact of hydrological changes on use of water for human consumption, for example, once again begins by tracing through chains of causality to estimate the changes in the quantity and quality of water available to consumers. The prices typically charged to consumers for this water, however, are not reliable measures of the value of the water to consumers, as they are set administratively, with no regard for supply and demand (indeed, in most cases water fees do not even cover the cost of delivering the water to consumers, let alone the value of the water itself). The value of an additional unit of water can be estimated in various ways, such as the cost of alternative sources of supply (see *cost-based measures*, below), or by asking consumers directly how much they would be willing to pay for it (see *contingent valuation*, below). Note that it's very important to use the value of an additional unit of water, since some amount of water is, of course, vital for survival. Thus an additional unit of water will be very valuable when water is scarce, but much less so when water is plentiful.

- When the impact is on water quality rather than quantity, the impact on well-being might be reflected in increased morbidity or even mortality. Again, the process begins by tracing through chains of causality, which in this case will include dose-response functions that tie concentrations of pollutants to human health. Valuing the impact on health itself can then be done in a number of ways (see *cost of illness and human capital*, below).

- In some cases, the impact is on relatively intangible aspects of well-being, such as aesthetic benefits or existence value. Particular efforts have been made in recent years to develop techniques to value such impacts, including *hedonic price*, *travel cost*, and *contingent valuation* methods (see below).

Cost of illness and human capital. The economic costs of an increase in morbidity due to increased pollution levels can be estimated using information on various costs associated with the increase in morbidity: any loss of earnings resulting from illness, medical costs such as for doctors, hospital visits or stays, medication, and any other related out-of-pocket expenses. The estimates obtained in this manner are interpreted as lower-bound estimates of the presumed costs or benefits of actions that result in changes in the level of morbidity, since this method disregards the affected individuals' preference for health versus illness, and restrictions on non-work activities. Also, the method assumes that individuals treat health as exogenous and does not recognize that individuals may undertake defensive actions (such as using special air or water filtration systems to reduce exposure to pollution) and incur costs to reduce health risks.

When this approach is extended to estimate the costs associated with pollution-related mortality (death), it is referred to as the *human-capital* approach. It is similar to the change-in-productivity approach in that it is based on a damage function relating pollution to productivity, except that in this case the loss in productivity of human beings, measured in terms of expected life-time earnings. Because it reduces the value of life to the present value of an individual's future income stream, the human-capital approach is extremely controversial when applied to mortality. Many economists prefer, therefore, not to use this approach and to simply measure the changes in the number of deaths (without monetary values) or measures such as Disability-Adjusted Life Years (DALYs) (see {section 3.3.4.1} below).

Cost-based approaches. The cost of replacing the services provided by the environmental resource can provide an order of magnitude estimate of the value of that resource. For example, if ecosystem change reduces the availability of drinking water, the cost of piping in water from an alternative source could be used. The major underlying assumptions of these approaches are (i) that the nature and extent of physical damage expected is predictable (there is an accurate damage function available), and (ii) that the costs to replace or restore damaged assets can be estimated with a reasonable degree of accuracy. It is further assumed that these costs can be used as a valid proxy for the cost of environmental damage. That is, the replacement or restoration costs are assumed not to exceed the economic value of the service. This assumption may not be valid in all cases. It simply may cost more to replace or restore a service than it was worth in the first place. Also, there may be more cost-effective ways to compensate for environmental damage than to replace the original service or restore it to its original condition, and these substitution possibilities are ignored with the use of this technique. If substitutes are available, the method will likely over-estimate the value of the service. Because of this, these methods are generally thought to provide an upper-bound estimate of value.

Hedonic analysis. The prices paid for goods or services that have environmental attributes differ depending on those attributes. Thus, a house in a clean environment will sell for more than an otherwise identical house in a polluted neighborhood. Hedonic price analysis compares the prices of similar goods to extract the implicit value that buyers place on the environmental attributes. However, this method requires a very large number of observations, and so its applicability is very limited.

Travel cost. The travel cost (TC) method is an example of a technique that attempts to deduce value from observed behavior in a surrogate market. It uses information on visitors' total expenditure to visit a site to derive their demand curve for the site's services. The technique assumes that changes in total travel costs are equivalent to changes in admission fees. From this demand curve, the total benefit visitors obtain can be calculated. (It is important to note that the value of the site is not given by the total travel cost; this information is only used to derive the demand curve.) This method was designed for and has been used extensively to value the benefits of recreation, but has limited utility in other settings.

Contingent valuation. Contingent valuation (CV) is an example of a stated preference technique. It is carried out by asking consumers directly about their willingness-to-pay to obtain an environmental service. A detailed description of the service involved is provided, along with details about how it will be provided. The actual valuation can be obtained in a number of ways, such as asking respondents to name a figure, having them choose from a number of options, or asking them whether they would pay a specific amount (in which case, follow-up questions with higher or lower amounts are often used). CV can, in principle, be used to value any environmental benefit, simply by phrasing the question appropriately. Moreover, since it is not limited to deducing preferences from available data, it can be targeted quite accurately to ask about the specific changes in benefits that the change in ecosystem condition would result in. Because of the need to describe in detail the good being valued, interviews in CV surveys are often quite time-consuming. It is also very important that the questionnaire be extensively pre-tested to avoid various sources of bias. CV methods have been the subject of severe criticism by some analysts. A "blue-ribbon" panel was organized by the US Department of Interior following controversy over the use of CV to value damages from the 1989 Exxon Valdez oil spill. The report of this panel (NOAA 1993) concluded that CV can provide useful and reliable information when used carefully, and provided guidance on doing so. This report is generally regarded as authoritative on appropriate use of the technique.

Choice modeling. Choice modeling (also referred to as Choice Experiments, Conjoint Analysis, or Attribute Based Stated Choice Method) is an alternative, newer approach to obtaining stated preferences. It consists of asking respondents to choose their preferred option from a set of alternatives where the alternatives are defined by attributes (including the price or payment). The alternatives are designed so that the respondent choice reveals their marginal rate of substitution between the attributes and money. These approaches are useful in cases in which the investigator is interested in the valuation of the attributes of the situation, or cases in which the decision lends itself to respondents choosing from a set of alternatives described by attributes. Advantages of choice modeling include: (1) the control of the stimuli is in the experimenter's hand, as opposed to the low level of control generated by real market data; (2) the control of the design yields greater statistical efficiency and eliminates collinearity; (3) the attribute range can be wider than found in market data; and (4) the introduction and/or removal of products, services and attributes is easily accomplished (Louviere et al. 2000; Holmes and Adamowicz 2003; Bateman et al. 2004). The disadvantages associated with the technique are that the responses are hypothetical and therefore suffer from problems of hypothetical bias (similar to contingent valuation) and that the choices can be quite complex when there are many attributes and alternatives. The econometric analysis of the data generated by choice modeling is also fairly complex.

Benefits transfer. A final category of approach is known as benefits transfer. This is not a methodology per se, but rather refers to the use of estimates obtained (by whatever method) in one context to estimate values in a different context. For example, an estimate of the benefit obtained by tourists viewing wildlife in one park might be used to estimate the benefit obtained from viewing wildlife in a different park. Benefits transfer has been the subject of considerable controversy in the economics literature, as it has often been used inappropriately. A consensus seems to be emerging that benefit transfer can provide valid and reliable estimates under certain conditions. These conditions include the requirement that the commodity or service being valued is identical at the site where the estimates were made and the site where they are applied; and that the populations affected have identical characteristics. Of course, the original estimates being transferred must themselves be reliable for any attempt at transfer to be meaningful.

Each of these approaches has seen extensive use in recent years, and an extensive literature exists on their application. These techniques can and have been applied to a very wide range of issues (McCracken and Abaza 2001), including the benefits of ecosystems such as forests (Bishop 1999), wetlands (Barbier et al. 1997; Heimlich et al. 1998), watersheds (Kaiser and Roumasset 2002), as well as ecosystems services such as water (Young and Haveman 1985), non-timber forest benefits (Lampietti and Dixon 1995; Bishop 1998), recreation (Bockstael et al. 1991; Herriges and Kling 1999), and cultural benefits (Pagiola 1996; Navrud and Ready 2002). Many valuation studies are cataloged in the Environmental Valuation Reference Inventory (EVRI) website, maintained by Environment Canada (EVRI 2004).

In general, measures based on observed behavior are always preferred to measures based on hypothetical behavior, and more direct measures are preferred to indirect measures. However, the choice of valuation technique in any given instance will be dictated by the characteristics of the case and by data availability. Several techniques have been specifically developed to cater to the characteristics of particular problems. The travel cost method, for example, was specifically developed to measure the utility derived by visitors to sites such as protected areas, and is of limited applicability outside that particular case. The change in productivity approach, on the other hand, is very broadly applicable to a wide range of issues. Contingent valuation is potentially applicable to any issue, simply by phrasing the questions appropriately, and as such has become very widely used – probably excessively so, as it is easy to mis-apply and, being based on hypothetical behavior, is inherently less reliable than measures based on observed behavior.

3.3.3.2 Putting Economic Valuation into Practice

Whatever valuation method is used, framing the question to be answered appropriately is critical. In most policy-relevant cases, the concern is over changes in the level and mix of services provided by an ecosystem. At any given time, an ecosystem provides a specific “flow” of services, depending on the type of ecosystem, its condition (the “stock” of the resource), how it is managed, and its socioeconomic context. A change in management (whether negative, such as deforestation, or positive, such as an improvement in logging practices) will change the condition of the ecosystem and hence the flow of benefits it is capable of generating. It is rare for all ecosystem services to be lost entirely; a forested watershed that is logged and converted to agriculture, for example, still provides a mix of provisioning, regulating, supporting, and cultural services, even though both the mix and the magnitude of specific services will have changed. The typical question being asked, then, is whether the total value of the mix of services provided by an ecosystem managed in one way is greater or smaller than the total value of the mix provided by that ecosystem if it were managed in another way. Consequently an assessment of this change in the value is typically most relevant to decisionmakers. Where the change does involve the complete elimination of ecosystem services, such as the conversion of an ecosystem through urban expansion or road-building, then the change in value would equal the total economic value of the services provided by the ecosystem. (Measurements of the total economic value of the services provided by an ecosystem can also be useful to policy-makers as an economic indicator, just as measures of gross domestic product or genuine savings provide policy-relevant information on the state of the economy.)

Assessing the change in value of the ecosystem services caused by a management change can be achieved either by explicitly estimating the change in value, or by separately estimating the value of ecosystem services under the current and the alternative management regime and then comparing them; if the loss of a given service is irreversible, then the loss of the option value of that service should also be included. (An important caveat here is that the appropriate comparison is that between the ecosystem with and without the management change; this is not the same as a comparison of the ecosystem before and after the management change, as many other factors will typically also have changed.)

The actual change in the value of the benefits can be expressed either as a change in the value of the annual flow of benefits, if these flows are relatively constant, or as a change in the present value of all future flows. The latter is equivalent to the change in the capital value of the ecosystem, and is particularly useful when future flows are likely to vary substantially over time. (It is important to bear in mind that the capital value of the ecosystem is not separate and additional to the value of the flows of benefits it generates;

rather, the two are intimately linked in that the capital value is the present value of all future flows of benefits.)

Estimating the change in the value of the flow of benefits provided by an ecosystem begins by estimating the change in the physical flow of benefits. This is illustrated in {Figure 3.3} for a hypothetical case of deforestation that affects the water services provided by a forest ecosystem. It is important to realize that the bulk of the work involved in the exercise actually concerns quantifying the biophysical relationships. In many cases, this requires tracing through and quantifying a chain of causality. Thus, valuing the change in production of irrigated agriculture resulting from deforestation requires (i) estimating the impact of deforestation on hydrological flows; (ii) determining how changes in water flows affect the availability of water to irrigation; and (iii) estimating how changes in water availability affects agricultural production. Only at the end of this chain does valuation in the strict sense occur – in putting a value on the change in agricultural production, which in this instance is likely to be quite simple as it is based on observed prices of crops and agricultural inputs. The change in value resulting from deforestation then requires summing across all the impacts. Clearly, tracing through these chains requires close collaboration between experts in different disciplines—in this example, between foresters, hydrologists, water engineers, and agronomists as well as economists. It is a common problem in valuation that information is only available on some of the links in the chain, and often in incompatible units. The MA can make a major contribution by making the various disciplines involved better aware of what is needed to ensure that their work can be combined with that of others to allow a full analysis of such problems.

In bringing the various strands of the analysis together, there are many possible pitfalls to be wary of. Inevitably, some types of value will prove impossible to estimate using any of the available techniques, either because of lack of data or because of the difficulty of extracting the desired information from them. To this extent, estimates of value will be under-estimates. Conversely, there is an opposite danger that benefits (even if accurately measured) might be double-counted.

As needed, the analysis can be carried out either from the perspective of society as a whole or from the perspective of individual groups within society. When the analysis is undertaken from the societal perspective, it should include *all* costs and benefits associated with ecosystem management decisions, which should be valued at their opportunity cost to society (sometimes known as “shadow prices”). In contrast, focusing on a particular group usually requires focusing on a subset of the benefits provided by an ecosystem, as that group may receive some benefits but not others; groups located within an ecosystem, for example, typically receive most of the direct use benefits but few of the indirect use benefits, whereas downstream users receive few direct use benefits but many indirect use benefits. It also requires using estimates of value specific to that group (the value of additional water, for example, will be different depending on if it is used for human consumption or for irrigation). The analysis can thus allow for distributional impacts and equity considerations to be taken into account, as well as overall impacts on well-being at the societal level. This type of disaggregation is also very useful to understand the incentives that particular groups face in making their ecosystem management decisions. Many ecosystems are mismanaged, from a social perspective, precisely because most groups that make decisions about ecosystem management perceive only a subset of the benefits it provides.

Assessing the impact of ecosystem change almost always requires comparing costs and benefits at different time. In economic analysis, this is achieved by *discounting* future costs and benefits so that all are expressed in today's monetary units (Portney and Weyant 1999). Because discounting makes future benefits appear smaller, this practice has been controversial, and some have called for use of lower (perhaps even zero) discount rate when assessing environmental issues. Discount rates, however, reflect preferences for current as opposed to future consumption. Whatever discount rate is chosen, it should be applied in all evaluations involving choices between outcomes occurring at different times.

Similarly, estimating the impact of changes in management on future flows of benefits allows for intergenerational considerations to be taken into account. Here too, the bulk of the work involved concerns predicting the change in future physical flows; the actual valuation in the narrow sense forms only a small part of the work. Predicting the value that future generations will place on a given service is obviously difficult. Technical, cultural, or other changes could result in the value currently placed on a service either increasing or decreasing. Often, the best that can be done is to simply assume that current values will remain unchanged. If trends suggest that a particular change in values will occur, that can be easily included in the analysis. Such predictions are notoriously unreliable, however.

3.3.4 Indicators of Specific Dimensions of Well-being

Well-being cannot be measured solely in terms of income, nor can non-income aspects of well-being always be measured in monetary terms. This section reviews several indicators that seek to capture specific aspects of well-being which economic valuation often captures imperfectly, if at all, including health, poverty, and vulnerability.

3.3.4.1 Health Indicators

Biological responses involved in human disease phenomena are among the most important set of parameters for assessing environmental quality, and measures in support of environmental protection are often justified on the basis of their impact on human health (Moghissi 1994).

Health indicators have been used extensively to monitor the health of populations and are usually defined in terms of health outcomes of interest. The majority of health indicators so far developed, however, have no direct reference to the environment; examples include simple measures of life expectancy, or cause-specific mortality rates where no attempt has been made to estimate these health outcomes attributable to the environment. An Environmental Health Indicator (EHI) can be seen as a measure that summarizes, in easily understandable and relevant terms, some aspect of the relationship between the environment and health that is amenable to action (Corvalan 1996). They are summarized measures (both of health outcomes and hazard exposures), which represent an underlying causal relationship between an environmental exposure and a health consequence (Pastides 1995). As with all indicators (see {section 3.2.6}), appropriate EHIs vary according to the problem and the context.

EHIs can be constructed by linking aggregate data (linkage-based), by identifying environmental indicators with a health linkage (exposure-based), or by identifying health indicators with an environmental linkage (outcome-based). There are special complexities in the identification of EHIs since the incidence of many environmentally related diseases cannot be easily traced back to specific environmental exposures (Kjellström 1995). The Driving forces-Pressure-State-Exposure-Effect-Action (DPSEEA) framework, proposed by the World Health Organization (WHO), is a widely accepted conceptual framework to guide the development of EHIs. The Driving Forces component refers to the factors that motivate and push the environmental processes involved (population growth; technological and economic development; policy intervention, etc.). The drivers result in the generation of pressures, normally expressed through human occupation or exploitation of the environment, and may be generated by all sectors of economic activity. In response to these pressures the state of the environment is often modified, producing hazards. Exposure refers to the intersection between people and the hazards in the environment. These exposures lead to a wide range of health effects, ranging from well-being through morbidity and/or mortality (Briggs 1999).

EHIs are needed to monitor both trends in the state of the environment and trends in health, resulting from exposures to environmental risk factors. They are useful also to compare areas or countries in terms of their environmental health status; to assess the effects of policies and other interventions on environmental health and also help to investigate potential links between environment and health (Briggs 1999). EHIs use a variety of units, but many are expressed in disability adjusted life years (DALYs): the sum of life years lost due to premature mortality and years lived with disability, adjusted for severity (Murray 1994; Murray 1997).

Usable EHIs depend heavily upon the existence of known and definable links between environment and health. Difficulties in establishing these relationships (due, for example, to the complexity of confounding effects and the problems of acquiring reliable exposure data) inhibit the practical use of many potential indicators and make it difficult to establish core indicators sets (Corvalan 2000). Thus, the presence of environmental changes does not translate automatically into health outcomes and the incidence of many environmentally-related diseases cannot be easily traced back to specific environmental exposures. Many broader environmental issues, such as deforestation, loss of biodiversity, soil degradation and climate change have a much less direct link to health. Although the effects of ecosystem disturbance on human health may be relatively direct they may also occur at the end of long, complex causal webs, dependent on many intermediate events. When these effects are subtle and indirect, often entailing complex interactions with social, conditions their measurement through indicators is often difficult.

The WHO, by assigning weight factors in the form of estimated environmental fraction to reported DALYs for relevant diseases, have estimated that 23% of the global burden of disease is related to environmental factors (WHO 1997).

Sets of specific EHIs have been proposed for the monitoring of both environmental quality and population health levels on a national basis, encompassing different types of hazards (chemical, physical, and biological) and modifications in several ecosystems, such as forests, agroecosystems, and urban ecosystems (Confalonieri 2001). In addition, indicators have recently been proposed to monitor the interactions between human health effects and the quality of specific ecosystems, including oceans (Dewailly 2002), freshwater ecosystems (Morris 2002), and urban systems (Hancock 2002). {Table 3.7} shows simple examples of how changes in ecosystem services generate hazards to human health and how these can be measured in the form of EHIs.

Table 3.7: Examples of ecosystem disruption and Environmental Health Indicators (EHIs)

Health Impact Assessment (HIA) provides a framework and a systematic procedure to estimate the health impact of a proposed intervention or policy action on the health of defined population groups. HIA produces hypothetical health trade-offs of adopting different courses of action (Scott-Samuel et al. 2001). These estimates may be converted in monetary values, to facilitate comparisons with non-health impacts. The procedure for applying an HIA typically involves a prospective assessment of a program or intervention before implementation, although it may be carried out concurrently or retrospectively. HIA gathers opinions and concerns regarding the proposed policy: and uses knowledge of health determinants as to the expected impacts of the proposed policy or intervention, and describes the expected health impacts using both quantitative and qualitative methods as appropriate.

3.3.4.2 Poverty and Equity

Possibly the most closely-watched impacts of ecosystem changes are those that pertain to poverty. Although poverty has historically been defined in strictly economic terms, in recent years a broader understanding of poverty has increasingly been used, in which poverty is understood as encompassing not only deprivation of materially-based well-being, but also a broader deprivation of opportunities (World Bank 2001). The MA conceptual framework recognizes five linked components of poverty: the necessary material for a good life, health, good social relations, security, and freedom and choice.

Despite the broader understanding of poverty, however, most poverty indicators pertain to monetary measures of well-being. Income has been most widely used as a poverty indicator. In recent years, however, many analysts have argued that consumption is a better measure, as it is more closely related to well-being and reflects capacity to meet basic needs through income and access to credit. It also avoids the problem of income flows being erratic at certain times of the year—especially in poor agrarian economies—which can cause reporting errors. Income-based poverty indicators, however, are easier to compare with other variables such as wages. They are also more widely collected, in contrast to consumption data that are seldom collected, thereby limiting the possibility of undertaking comparative analyses.

Monetary-based indicators have the further limitation that they cannot reflect individuals' feeling of well-being and their access to basic services. A household's ability to address risks and threats (and hence, its feeling of well-being) can change dramatically even as income and consumption remain stable. Factoring in the effect of vulnerability, analysts estimate that monetary-based indicators can understate poverty and inequality by around 25 percent (World Bank 2001). In response, efforts have been made to develop non-monetary based poverty indicators such as outcomes relating to health, nutrition, or education, as well as composite indices of wealth (Wodon and Gacitúa-Marió 2001). These alternative poverty indicators, however, face methodological and data collection issues that make comparisons between countries difficult.

Poverty measures are defined relative to a poverty line (the cut-off separating the poor from the non-poor). Many types of poverty measures exist, but the most commonly used are the headcount index (a measure of poverty incidence, which computes the number of people or share of the population below the poverty line), the poverty gap (a measure of the depth of poverty, which describes how far below the poverty line people are), and the squared poverty gap (a measure of poverty severity, which combines both poverty gap and inequality among the poor). A related set of measures is used to measure inequality, including the Gini coefficient (a measure between 0 and 1 with 0 representing perfect equality and 1 perfect inequality) and the Atkinson index (which incorporates the strength of societal preference for equality).

Most countries determine their own poverty line, making international comparisons of poverty data conceptually and practically difficult. Poverty lines in rich countries are characterized by a higher purchasing power than in poorer nations, making comparisons subject to possible inaccurate interpretation (World_Bank 2003). In response, an international poverty line was established in order to measure poverty across countries. The dollar-a-day poverty line (this has been updated to $1.08 a day in 1993 prices) was chosen. It is converted to local currency units using the purchasing power parity (PPP) exchange rates. However, the non-uniform derivation of the PPP changes the relative value of expenditures between countries, and may affect poverty comparisons. As such the World Bank, for example, uses the PPP-based international poverty line to arrive at comparable aggregate poverty estimates across countries, but relies mostly on national poverty lines in its poverty analysis.

Reliable and consistent poverty analyses require uniform and high-quality data that are in many cases—especially in developing countries—not available. The Living Standards Measurement Study (LSMS) program was established to develop methods to monitor progress in improving standards of living, in identifying the impacts of policy reforms on well-being, and in establishing a common language by which research proponents and policy makers can communicate (Grosh and Glewwe 1995). LSMS surveys are used to gather data on a gamut of household activities many of which are used as poverty indicators. Well-being is measured by consumption and hence in most LSMS research on poverty, measurement of consumption is heavily emphasized in the surveys. With the strong interest on addressing poverty issues in the context of sustainable development, there are current efforts to expand the scope of the LSMS surveys to include variables pertaining to natural resource and environmental management. Exploratory efforts are being undertaken to possibly include a module on environmental health in the LSMS research.

The link between poverty and ecosystem services is established by monitoring ecosystem changes and observing how they change poverty measures. Whether the poor are agents or victims of environmental degradation (or both), and the issue of possible trade-off between ecosystem condition and the well-being of the poor are burning topics among scholars and policy makers (Reardon and Vosti 1997; World_Bank 2002). Recent work has documented that the poor tend to rely heavily on goods and services provided by the environment, and thus are particularly vulnerable to their degradation (Cavendish 1999).

3.3.4.3 Other Indicators

A great number of other indicators can be used to assess various dimensions of human well-being. For example, several indicators exist that can help measure progress towards achieving the Millennium Development Goals (in addition to the poverty and health indicators described above) (World Bank 2002). Adult literacy rates measure educational attainment, and indicators such as net enrollment ratios in primary education or the proportion of students who start grade 1 who reach grade 5 can measure progress towards the goal of universal primary education (goal 2). The ratio of girls to boys at various levels of education, the ratio of literate females to males, the share of women in non-agricultural employment, and the share of seats in parliament held by women can be used to measure progress towards the goal of promoting gender equality (goal 3), and maternal mortality ratios and the proportion of births attended by skilled personnel can be used to measure progress towards improving maternal health (goal 5). These, and many other, indicators can provide valuable insights, but they are often difficult to relate to ecosystem condition as they are also affected by many other factors. (Risk and vulnerability indicators are discussed in chapter {VULNERABILITY CHAPTER}.)

3.3.5 Aggregate Indicators of Human Well-being

Several indicators are in use as aggregate indicators of human well-being. The most commonly used, of course, is the Gross Domestic Product (GDP), which is a measure of economic activity. This indicator has long been known to be imperfect, even for the narrow purpose of measuring economic activity, let alone as a measure of overall well-being. The limitations of GDP as an indicator have led to substantial efforts to improve it, and to develop alternative indicators.

The linkage between human well-being and national accounting is not particularly straightforward, since Gross Domestic Product (GDP), for example, includes both consumption of produced goods—yielding direct benefits for well-being—and investment in physical capital. Moreover many factors, including the enjoyment of environmental amenities, are not captured in the value of consumption recorded in the national accounts. Recent results in the theory of environmental accounting make the linkage between asset accounting and well-being explicit. Hamilton and Clemens (1999) show that there is a direct link between the change in the value of all assets (including produced and natural assets) and the present value of social utility (or well-being): declining asset values, measured at current shadow prices, imply future declines in social well-being. Dasgpupta and Maler (2000) and Asheim and Weitzman (2001) have extended these results. The World Bank has been publishing estimates of net or 'genuine' saving for roughly 150 countries since 1999 (World_Bank 2003). Relying on internationally available data sets, these estimates adjust traditional measures of saving to reflect investments in human capital, depreciation of produced capital, depletion of minerals, energy and forests, and damages from emissions of CO_2.

Efforts to develop alternative indicators of well-being include efforts to develop composite indices that capture the multi-dimensionality of well-being. Early attempts to develop composite indices include the Weighted Index of Social Progress (WISP) (Estes 1984; 1988) and the Physical Quality of Life Index (PQLI) (Morris 1979). More recently, the Human Development Index (HDI) (UNDP 1998; 2003), which combines measures of life expectancy, literacy, education enrollment, and GDP per capita, has been widely used. The Human Poverty Index (HPI) is similar, but with different variables for developed and developing countries, while the gender-related development index (GDI) adjusts for disparities in achievement for men and women (UNDP 2003). None of these indicators include environmental variables explicitly. One indicator that does is the Calvert-Henderson Quality of Life Indicator, which includes measures of environmental, social, and economic conditions (Flynn 2000; Henderson 2000). Composite indicators suffer from the arbitrariness of the weighting of their different components, however. Some prefer to simply list the components individually, without attempting to aggregate them into a single measure. Thus the World Bank provides a wide panoply of indicators in their annual *World Development Indicators* (WDI) publication (World_Bank 2004), and UNDP provide a variety of indicators in addition to the aggregated HDI in their *Human Development Report* (UNDP 2003). All of these indicators have substantial limitations from the perspective of the MA, as they are extremely difficult to relate to environmental conditions.

3.3.6 Intrinsic Value

Economic valuation attempts to measure the utilitarian benefits provided by ecosystems. In addition, many people ascribe ecological, sociocultural, or intrinsic values to the existence of ecosystems and species and, sometimes, inanimate objects such as “sacred” mountains.

Some natural scientists have articulated a theory of value of ecosystems in reference to the causal relationships between parts of a system—for example, the value of a particular tree species to control erosion or the value of one species to the survival of another species or of an entire ecosystem (Farber et al. 2002). At a global scale, different ecosystems and their species play different roles in the maintenance of essential life support processes (such as energy conversion, biogeochemical cycling, and evolution). The magnitude of this ecological value is expressed through indicators such as species diversity, rarity, ecosystem integrity (health), and resilience. The concept of ecological value is captured largely in the “supporting” aspect of the MA’s definition of ecosystem services.

What might be called sociocultural value derives from the value people place on elements in their environment based on different worldviews or conceptions of nature and society that are ethical, religious, cultural, and philosophical. A particular mountain, forest, or watershed may, for example, have been the site of an important event in their past, the home or shrine of a deity, the place of a moment of moral transformation, or the embodiment of national ideals. These values are expressed through, for example, designation of sacred species or places, development of social rules concerning ecosystem use (for instance, “taboos”), and inspirational experiences. For many people, sociocultural identity is in part constituted by the ecosystems in which they live and on which they depend—these help determine not only how they live, but who they are. To some extent this kind of value is captured in the concept of “cultural” ecosystem services and can be valued using economic valuation techniques. To the extent, however, that ecosystems are tied up with the very identity of a community, the sociocultural value of ecosystems transcends utilitarian preference satisfaction. These values might be elicited by using, for example, techniques of participatory assessment (Campell and Luckert 2002).

The notion that ecosystems have intrinsic value is based on a variety of points of view. Intrinsic value is a basic and general concept that is founded upon many and diverse cultural and religious worldviews. Among these are indigenous North and South American, African, and Australian cultural worldviews, as well as the major religious traditions of Europe, the Middle East, and Asia. In the Judeo-Christian-Islamic tradition of religions, human beings are attributed intrinsic value on the basis of having been created in the image of God. Some commentators have argued that plant and animal species, having also been created by God and declared to be “good,” also have intrinsic value on the same basis (Barr 1972; Zaidi 1981; Ehrenfeld and Bently 1985). In some American Indian cultural worldviews, animals, plants, and other aspects of nature are conceived as relatives, born of one universal Mother Earth and Father Sky (Hughes 1983). The essential oneness of all being, *Brahman*, which lies at the core of all natural things, is basic to Hindu religious belief (Deutch 1970). Closely related to this idea is the moral imperative of *ahimsa,* non-injury, extended to all living beings. The concept of *ahimsa* is also central to the Jain environmental ethic (Chapple 1986). In democratic societies the modern social domain for the ascription of intrinsic value is the parliament or legislature (Sagoff 1998). In other societies a sovereign power ascribes intrinsic value, although this may less accurately reflect the actual values of citizens than parliamentary or legislative acts and regulations do. The metric for assessing intrinsic value is the severity of the social and legal consequences for violating laws prohibiting a market in or otherwise compromising that which is recognized to be intrinsically valuable.

3.4 Assessing Trade-offs in Ecosystem Services

The challenge to decision-making is to make effective use of new information and tools in this changing context in order to improve the decisions that intend to enhance human well-being and provide for a sustainable flow of ecosystem services. Perhaps the most important traditional challenge in decision-making about ecosystems is the complex tradeoff faced when making decisions about how to alter ecosystems. Increasing the flow of one service from a system, such as provision of timber, may decrease the flow from others, such as carbon sequestration or the provision of habitat. In addition, benefits, costs, and risk are not allocated equally to everyone, so any intervention will change the distribution of human well-being—another trade-off. A crucial issue for the MA is to provide information for assessing the trade-offs among ecosystem services resulting from policy decisions.

Understanding the impact of ecosystem management decisions would be simplest if all impacts were expressed in common units. If information on the impact of ecosystem change is presented solely as a list of consequences in physical terms—so much less provision of clean water, perhaps, and so much more production of crops—then the classic problem of comparing apples and oranges applies.

The purpose of economic valuation is to make the disparate services provided by ecosystems comparable to each other, by measuring their relative contribution to human well-being. As utility cannot be measured directly, economic valuation usually attempts to measure all services in monetary terms. This is purely a matter of convenience, in that it uses units that are widely recognized, saves the effort of having to convert values already expressed in monetary terms into some other unit, and facilitates comparison with other activities that also contribute to well-being, such as spending on education or health. In particular, it expresses the impacts of ecosystem change into units that are readily understood by decisionmakers and the lay public alike. When all impacts of ecosystem change are expressed in these terms, then they can readily be introduced into frameworks such as cost-benefit analysis in order to assess policy alternatives.

Other metrics are occasionally proposed. For example, some have advocated the use of energy units (Odum and Odum 1981; Hall et al. 1986), arguing that as all goods and services are ultimately derived from natural resources by expending energy, energy is the real source of material wealth. These approaches can provide valuable insights into particular issues. For purposes such as that of the MA, however, these approaches have several disadvantages – in particular, they have no direct link to human well-being, and they require a considerable effort to convert a wide variety of impacts into common units.

Efforts to place everything into common units will necessarily remain incomplete, however, sometimes because of lack of data, and sometimes because value arises not from utilitarian benefits but from intrinsic value or from another source of value. Societies have many objectives, only some of them purely utilitarian. Furthermore, the value of an ecosystem service varies depending on whether a critical threshold for ecosystem condition or human well-being is crossed (Farber et al. 2002). In other words, placing everything into common units is sometimes impossible, and frequently undesirable. It is important to stress, however, that even incomplete efforts to express impacts in common units can be helpful, by reducing the number of different dimensions that need to be taken into considerations.

Graphical depictions of the trade-offs in ecosystem services associated with alternative policy options can provide useful input to decision makers. "Spider diagrams" such as that in {Figure 3.4} can depict the amount of ecosystem services associated with different management alternatives. For example, {Figure 3.4} depicts hypothetical trade-offs among five ecosystem services associated with an expansion of cropland in a forested area: food production, carbon sequestration, species richness, soil nutrients, and base streamflow. Comparison of the ecosystem services available before forest conversion to cropland {Figure 3.4a} with the services after forest conversion {Figure 3.4b} allows a decision maker to account for the full suite of ecosystem services affected by the conversion. The approach requires quantifiable and measurable indicators for each of the services depicted. The quantities depicted can be an absolute measure (e.g., tonnes of carbon stored), relative to a previous quantity, to a relevant average quantity (e.g., for the area, or for the biome), or to an ideal "sustainable" amount. The degree to which the diagram effectively communicates trade-offs in ecosystem services depends on the explicit definition of the values on the axes and the ability to quantify them. A series of diagrams, for varying time since clearing and for varying spatial scales of interest, could be used to inform decision makers about the effects on ecosystem services for the varying scales of analysis. When a large number of management alternatives are to be compared, they can be portrayed either in a series of spider diagrams or compared across all management alternatives as in {Figure 3.5} (Heal et al. 2001a).

Figure 3.4: Spider diagram used to depict hypothetical trade-offs in a policy decision to expand cropland in a forested area.

Figure 3.5: Portrayal of hypothetical trade-offs in ecosystem services associated with management alternatives for expanding cropland in a forested area.

Depictions of ecosystem services associated with pre-defined management alternatives, as in {Figures 3.4 and 3.5}, are simple and readily communicable to decision-makers but are often unable to account for non-linearities and thresholds in responses of ecosystem services to management decisions. When such phenomena are present, figures such as {Figure 3.6} can be useful to asses choices. For example, application of nitrogen fertilizer involves a trade-off between increasing crop yields and decreasing coastal fisheries if nitrate leaching leads to hypoxia in downstream coastal locations, such as in the Mississippi Delta (Donner et al. 2002; Donner et al. submitted). Balancing an objective of maximum crop yields with minimum damage to coastal fisheries requires knowledge of the response curves of each service to nitrogen fertilizer application {Figure 3.6}. In this example, fertilizer application beyond point “A” results in negligible increase in crop yield but substantial nitrate leaching. A decision to apply fertilizer greater than point “A” trades small increases in crop yield for large increases in nitrate leaching. A decision to apply fertilizer less than point “A” trades small decreases in nitrate leaching for foregone large increases in crop yield. To the extent that the shape of the response curves can be quantified, management alternatives can account for these types of non-linear responses to determine the most desirable alternative.

Figure 3.6: Example of non-linear responses of two ecosystem services (crop yields and coastal fisheries) to application of nitrogen fertilizer.

Portraying interactions among multiple ecosystem services graphically quickly becomes complex and unwieldy. Heal *et al*. (2001a) suggest constructing “production possibility frontiers” to model combinations in the amounts of ecosystem services possible to achieve a management objective. For example, possible combinations of ecosystem services such as carbon storage and timber production can be modeled to achieve varying levels of water purification. The optimal mix of these services can then be selected depending on the management objectives

Multi-criteria analyis provides another formal framework to help assess choices in the presence of multiple, perhaps contradictory, objectives (Falconí 2003). In a multicriteria analysis, a matrix is constructed showing how each of the alternatives, under consideration ranks relative to the other alternatives, according to each criteria. This impact matrix, which may include quantitative, qualitative, or both types of information, allows the best alternative to the decision or analysis problem to be found (Munda 1995; Martínez-Alier et al. 1998). A vast number of multicriteria methods have been developed and applied for different policy purposes in different contexts (Munda 1995). The main advantage of multicriteria models is that they make it possible to consider a large number of data, relations, and objectives that are generally present in a specific real-world decision problem, so that the decision problem at hand can be studied in a multidimensional fashion. However, when different conflicting evaluations are taken into consideration, a multicriteria problem is mathematically ill defined. The application of the different methods can lead to different solutions and in some cases, solutions that satisfy multiple objectives may not be possible.

Consideration of the trade-offs involves clear definitions about the spatial and temporal scales of interest. How are future impacts on ecosystem services included in the analysis? Over what time frame should these impacts be considered? Does the alteration in ecosystem services affect human well-being distant in space from the ecosystem change (e.g., through downstream effects or atmospheric transport)? How are impacts that cross administrative or ecosystem boundaries incorporated in the analysis? Assessments need to be conducted within a scale domain appropriate to the processes or phenomena being examined. Cost-benefit analysis has often fallen short in the past in part because the spatial and temporal boundaries it used did not encompass all the impacts of the proposed interventions (Dixon *et al.*, 1994). This same weakness applies to all assessment methodologies: they will only be meaningful if the spatial and temporal scales of the analysis have been carefully defined. Too narrow a definition of either could result in a misperception of the problems (for example, if soil nutrients decline over time under agricultural use, the perceived impact on that dimension depends crucially on the time period chosen for the indicators shown in the second diagram).

3.5 Summary and Conclusions

This chapter describes the overall analytical approach used in this report to assess conditions and trends in ecosystems, the services they provide, and the implications for human well-being. The analytical approach aims to provide scientifically-based input to policy decisions that affect multiple ecosystem services, either as an intentional result of the policy (e.g., timber production) or as an unintended consequence (e.g., habitat loss). Accounting for these trade-offs involves quantifying the effects of the management decision on ecosystem services and human-well being in comparable units over explicitly-defined spatial and temporal scales.

Rigorous analysis involves quantifying implications of changes in ecosystem condition (e.g., forest conversion to cropland) for ecosystem services (e.g., flood protection) and effects on human well-being (e.g., damage from downstream flooding). The availability and accuracy of data sources and methods vary for different ecosystem services and different ecosystems around the world. These variations are reflected in the individual chapters of this report. Data on trends in "provisioning" services are more readily available than for "cultural", "supporting", and "regulating" services. Methods to quantify changes in ecosystem condition (e.g., remote sensing to determine forest area) are more mature than methods to quantify the effects on ecosystem services. Even more uncertain are methods to link changes in ecosystem services with aspects of human well-being not captured through economic valuation techniques (e.g., change in human well-being through loss of cultural and spiritual ecosystem services).

Data sources and methods used in this report were generally not developed explicitly for the assessment carried out in this report. However, the combination of approaches – including computer modeling, natural resource and biodiversity inventories, remote sensing and geographic information systems, traditional knowledge, case studies, indicators of ecosystem conditions and human well-being, and economic valuation techniques – provides a strong scientific foundation for the assessment. Systematic data collection for carefully-selected indicators reflecting trends in ecosystem condition and their services would provide a basis for future assessments.

Appendix. Core datasets used by the MA to assess ecosystem conditions and trends

Data and the MA

The Millennium Assessment has involved the development and distribution of a range of datasets and indicators. Although the overall MA products primarily consist of syntheses of findings from existing literature, the data and indicators developed or presented within the MA play important roles both in presenting information on the links between ecosystems and human well-being and in establishing a year 2000 'baseline' conditions for reference in future global and sub-global assessments.

For many central themes of the MA, there are multiple available datasets on which elements of the assessment could be based, and from which different conclusions could be drawn. For example, there is a range of land cover datasets available, based on information from different satellite sensors (see Table 3.2) and interpretation techniques, and from which different statistics on land cover could be generated. To ensure consistency of analysis and comparability of results across the chapters and working groups of the MA, a small number of MA "core datasets" have been selected (see {Table 3.8}). Although chapter teams are also making use of alternative datasets, applicable findings will in each case also be presented based on an analysis with the various core datasets, and an assessment conducted of the strengths and weaknesses of these datasets for the particular application in the chapters.

Table 3.8: Summary of MA Core Datasets

A discussion of the choice of MA systems, the main reporting unit for the Condition and Trends Working Group, can be found in {section 1.4.3 of chapter 1}. {Table 3.9} presents the updated system boundary definitions, adding detail to the brief system descriptions given in {Box 3} of {chapter 2}.

Table 3.9: MA System boundary definitions

MA Data Management

Data management procedures have been developed for the use of datasets in the MA. A web-based data catalogue will record metadata for all datasets used in the MA, and will be populated during 2004, as data use is finalized. Data Archives will be established at CIESEN and UNEP–WCMC, for all data in categories 4-6 of {Table 3.10}, as well as some data in category 2 if it is used for a significant portion of analysis in a particular chapter. MA archived data will be made freely accessible to any user, and all archived datasets will be accompanied by the draft ISO metadata standard (ISO 19115: Geographic Information).

Table 3.10:Data handling procedures in the MA

Abbreviations/Glossary

{Incomplete; Need to decide which of these need a glossary definition}

AVHRR	Advanced Very High Resolution Radiometer
CV	Contingent Valuation
DALY	Disability Adjusted Life Year
DPSEEA	Driving forces-Pressure-State-Exposure-Effect-Action
EHI	Environmental Health Indicator
FAO	United Nations Food and Agriculture Organization
FPAR	
GIS	Geographic Information System
GDI	Gender-related Development Index
GDP	Gross Domestic Product
GPS	Global Positioning System
HDI	Human Development Indicator
HPI	Human Poverty Index
IBI	Index of Biotic Integrity
IUCN	World Conservation Union
LAI	Leaf Area Index
LSMS	Living Standards Measurement Study
NOAA	National Oceanographic and Atmospheric Administration
PPP	Purchasing Power Parity
PQLI	Physical Quality of Life Index
PRA	Participatory Rural Appraisal
PVA	Population Viability Analysis
RRA	Rapid Rural Appraisal
TC	Travel Cost
TEK	Traditional ecological knowledge
UNEP	United Nations Environment Programme
WHO	World Health Organization
WISP	Weighted Index of Social Progress
WTA	Willingness to accept compensation
WTP	Willingness to pay

µm
pixel

Meta-analysis

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Table 3.1: Data sources and analytical approaches for assessing ecosystem conditions and trends

	Data source/analytical method						
Type of information required	Remote sensing and GIS	Natural resource and biodiveristy inventories inventories	Socioeconomic data	Ecosystem models	Indicators of ecosystem condition	Indigenous and traditional knowledge	Case studies of ecosystem response to drivers
Current spatial extent and condition of ecosystem	X	X			X		
Quality, quantity, and spatial distributions of services provided by system		X		X			
Human populations residing in and deriving livelihoods from system			X			X	X
Trends in ecosystem conditions and services	X	X		X	X	X	X
Response of ecosystem condition and services to drivers				X	X	X	X

Table 3.2: Satellite sensors for monitoring land cover, land surface properties, and land and marine productivity

Platform	***Sensor***	***Spatial resolution at nadir***	***Date of observations***
Coarse Resolution Satellite Sensors (> 1 km):			
NOAA-TIROS (National Oceanic and Atmospheric Administration- Television and Infrared Observation Satellite)	AVHRR (Advanced Very High Resolution Radiometer)	1.1km (Local Area Coverage) 8km (Global Area Coverage)	1978 – present
SPOT (Satellite Probatoire d'Observation)	VEGETATION	1.15km	1998 – present
ADEOS-II (Advanced Earth Observing Satellite)	POLDER (Polarization and Directionality of the Earth's Reflectances	7km x 6km	2002 –present
SeaStar	SeaWIFS (Sea viewing Wide Field of View)	1km (local coverage); 4km (global coverage)	1997 – present

1

Moderate Resolution Satellite Sensors (250 m - 1 km):			
ADEOS-II (Advanced Earth Observing Satellite)	GLI (Global Imager)	250m-1km	2002 – present
EOS AM and PM (Earth Observing System)	MODIS (Moderate Resolution Spectroradiometer)	250-1000m	1999 – present
EOS AM and PM (Earth Observing System)	MISR (Multi-angle Imaging Spectroradiometer)	275m	1999 – present
Envisat	MERIS (Medium resolution imaging spectroradiometer)	350-1200m	2002 – present
Envisat	ASAR (Advanced Synthetic Aperature Radar)	150-1000m	2002 – present
High Resolution Satellite Sensors (250 m - 1 km):			
SPOT (Satellite Probatoire d'Observation)	HRV (High Resolution Visible Imaging System)	20m; 10m (panchromatic)	1986 – present
ERS (European Remote Sensing Satellite)	SAR (Synthetic Aperture Radar)	30m	1995 - present
Radarsat		10-100m	1995 – present
Landsat (Land Satellite)	MSS (Multispectral Scanner)	83m	1972 - 1997
Landsat (Land Satellite)	TM (Thematic Mapper)	30m (120m thermal-infrared band)	1984 – present
Landsat (Land Satellite)	ETM+ (Enhanced Thematic Mapper)	30m	1999 – present
EOS AM and PM (Earth Observing System)	ASTER (Advanced Spaceborne Thermal Emission and Reflection Radiometer)	15-90m	1999 – present
Very High Resolution Satellite Sensors (< 20 m):			
JERS (Japanese Earth Resources Satellite)	SAR (Synthetic Aperature Radar)	18m	1992 - 1998
JERS (Japanese Earth Resources Satellite)	OPS	18mx24m	1992 - 1998
IKONOS		1m panchromatic; 4m multispectral	1999 - present
QuickBird		0.61m panchromatic; 2.44m multispectral	2001 - present

Note: The list is not intended to be comprehensive.

Table 3.3: Examples of resource inventories applicable for assessing ecosystem condition and trends

TYPE	SOURCE	DESCRIPTION
FOREST RESOURCES:		
Forest area and change	Food and Agriculture Organization of the United Nations (FAO). 2001. Global Forest Resources Assessment 2000 Main Report. FAO Forestry Paper 140. Rome: FAO.	Published every ten years (1980, 1990, 2000). Provides national and global level estimates of total forest area and net changes during the preceding decade, as well as information on plantations, forest ownership, management, and environmental parameters such as forest fires and biomass volumes.
Forest products	Food and Agriculture Organization of the United Nations (FAO). 2001. State of the World's Forests 2001. Rome: FAO.	Published every two years. Provide summary tables of national and regional production statistics for major categories of industrial roundwood, pulp and paper.
	International Tropical Timber Organization (ITTO). Annual Review and Assessment of the World Timber Situation 2001. Document GI-7/01. Yokohama, Japan: ITTO.	Published annually. Tabular databases on volume and value of production, consumption, and trade among ITTO producer and consumer countries. Time series for 5 years prior to publication.
Wood energy	International Energy Agency (IEA). 2002. Energy Statistics and Balances of OECD and Non-OECD Countries, 1999-2000. Paris: IEA. (4 reports.)	Published every two years. IEA data since 1994-95 have covered combustible renewables and waste in national energy balances, including disaggregated data for production and consumption of wood, charcoal, black liquor, and other biomass. Data provided at national and various regional aggregate levels.
AGRICULTURAL RESOURCES:		
Agricultural land, products, and yields	FAOSTAT-Agriculture. Data available on-line	Time series data since 1961 on extent of agricultural land use by country and region, production of primary and processed crops, live animals, primary and processed animal products, imports, exports, food balance sheets, agricultural inputs, and nutritional yield of many agricultural products.
Specific products	Member organizations of the Consultative Group on International Agricultural Research (CGIAR).	Issue-specific datasets on crops, animals, animal products, agricultural inputs, and genetic resources. Variety of spatial and temporal scales.

FISH RESOURCES:		
Fish stocks	Food and Agriculture Organization of the United Nations (FAO). 1997. Review of the State of World Fishery Resources: Marine Fisheries. FAO Fisheries Circular No. 920 FIRM/C920(En). Rome: FAO.	Tabular information on the state of exploitation, total production and nominal catches by selected species groups for major world fisheries.
Marine and inland fisheries	Food and Agriculture Organization of the United Nations (FAO). FISHSTAT. Data available on-line at http://www.fao.org/fi/statist/statist.asp.	Databases on fishery production from marine capture and aquaculture, fish commodity production and trade. Global, regional and national level data. Time series range from 20 to 50 years.
	Food and Agriculture Organization of the United Nations (FAO). 2002. The State of World Fisheries and Aquaculture 2002. Rome: FAO.	Published every two years. Data on five-year trends in fisheries production, utilization and trade for the world, geographic and economic regions. National level data for major fishing countries. Also provides extensive analysis of fishery issues.
	Food and Agriculture Organization of the United Nations (FAO). 2000. Yearbook of Fishery Statistics.	Updated annually. Includes aquaculture production, capture production, by country, fishing area, principal producers and principal species. Also trade data in fishery products.
	Food and Agriculture Organization of the United Nations (FAO). Fisheries Global Information System (FIGIS) at http://www.fao.org/fi/figis	Information on aquatic species, marine fisheries, fisheries issues and, under development in collaboration with regional fishery bodies, the state of marine resources and inventories of fisheries and fishery resources
	International Center for Living Aquatic Management (ICLARM). FishBase 2000.	Database on more than 27,000 fish species and references. Many datasets incomplete.
FRESHWATER/INLAND WATER RESOURCES:		
	Food and Agriculture Organization of the United Nations (FAO). AQUASTAT.	Global data on water resources and irrigation by country and region. Information on average precipitation, total internal water resources, renewable groundwater and surface water, total renewable water resources and total exploitable water resources.
	State Hydrological Institute, St. Petersburg, Russia. Igor A. Shiklomanov; UNESCO, Paris. World Water Resources and Their Use.	Global database on surface water resources and sectoral use. Includes water use forecasts to 2025.

Table 3.4: Examples of numerical models for assessing conditions and trends in ecosystems and their services

Type of Model	*Description*	*Examples of models*
Climate and land-atmosphere models	Land surface models of exchanges of water, energy, and momentum between land surface and atmosphere	(Sellers 1986); (Liang 1996)
Watershed and hydrologic models	Large basin models of hydrologic processes and biogeochemical exchanges in watersheds	(Fekete et al. 2002); (Green et al. in press); (Seitzinger and Kroeze 1998)
Population and metapopulation models	Models of dynamics of single populations predicting future abundance and trends, risk of decline or extinction, and chance of growth. They can be scalar, structured (e.g., age-, stage-, and/or sex-based), or individual-based, and incorporate variability, density-dependence, and genetics. Metapopulation models focus on the dynamics of, and interactions among, multiple populations, incorporating spatial structure and dispersal and internal dynamics of each population. Their spatial structure can be based on the distribution and suitability of habitat, and they can be used to assess species extinction risks and recovery chances.	(Akçakaya 2002); (Lacy 1993)
Community or food-web models	Models focusing on the interactions among different trophic levels (producers, herbivores, carnivores) or different species (e.g., predator-prey models).	(Park 1998); (USDA 1999);
Ecosystem process models	Models that include both biotic and abiotic components, and represent physical, chemical, and biological processes in coastal, freshwater, marine, or terrestrial systems. They can predict, for example, vegetation dynamics including temporal changes in forest species and age structure.	(Pastorok 2002)
Global terrestrial ecosystem models	Models of biogeochemical cycling of carbon, nitrogen, and other elements between the atmosphere and biosphere at the global scale, including vegetation dynamics, productivity, and response to climate variability.	(Field 1995; Foley 1996; McGuire 2001; Sitch et al. 2003)
Multi-agent models	Agents are represented by rules for behavior based on interactions with other actors or physical processes.	(Moss et al. 2001)
Integrated assessment models	Models that assemble, summarize, and interpret information to communicate to decision-makers	(Alcamo 1994)

Table 3.5: Examples of indicators to assess ecosystem condition and trends

Characteristic described by indicator	Example of Indicator	Availability of data for indicator	Units
Direct drivers of change:			
Land cover conversion	Area undergoing urbanization	high	hectares
Invasive species	Native vs. non-native species	medium	% of plant species
Climate change	Annual rainfall	high	mm/yr
Irrigation	Water usage	high	ft3/yr
Ecosystem condition:			
Condition of vegetation	Landscape fragmentation	medium	mean patch size
Condition of soil	Soil nutrients	medium	Nutrient concentration
	Soil salinization	low	Salt concentration
Condition of biodiversity	Species richness	low	No. of species/unit area
	Threatened species	medium	% of species at risk
	Viability of indicator species	low-medium	Probability of extinction
Condition of freshwater	Presence of contaminants	high	Concentration of pollutants Index of Biotic Integrity
Ecosystem service:			
Production service	Food production	high	Yield (kg/ha/yr)
Capacity to mitigate floods	Change in stream flow per unit precipitation	low	Discharge (m3/sec)
Capacity for cultural services	Spiritual value	low	?
Capacity to provide biological products	Biological products of potential value	low	Number of products or economic value

Note: See {section 3.3.4} for indicators of human well-being.

Table 3.6: Main economic valuation techniques

Methodology	Approach	Applications	Data requirements	Limitations
Change in productivity	Trace impact of change in environmental services on produced goods	Any impact that affects produced goods	Change in service; impact on production; net value of produced goods	Data on change in service and consequent impact on production often lacking
Cost of illness, human capital	Trace impact of change in environmental services on morbidity and mortality	Any impact that affects health (e.g., air or water pollution)	Change in service; impact on health (dose-response functions); cost of illness or value of life	Dose-response functions linking environmental conditions to health often lacking; under-estimates, as omits preferences for health; value of life cannot be estimated
Replacement cost (and variants, such as relocation cost)	Use cost of replacing the lost good or service	Any loss of goods or services	Extent of loss of goods or services, cost of replacing them	Tends to over-estimate actual value
Travel cost (TCM)	Derive demand curve from data on actual travel costs	Recreation	Survey to collect monetary and time costs of travel to destination, distance traveled	Limited to recreational benefits; hard to use when trips are to multiple destinations
Contingent valuation (CV)	Ask respondents directly their WTP for a specified service	Any service	Survey that presents scenario and elicits WTP for specified service	Many potential sources of bias in responses; guidelines exist for reliable application
Hedonic prices	Extract effect of environmental factors on price of goods that include those factors	Air quality, scenic beauty, cultural benefits	Prices and characteristics of goods	Requires vast quantities of data; very sensitive to specification
Benefits transfer	Use results obtained in one context in a different context	Any for which suitable comparison studies are available	Valuation exercises at another, similar site	Can be wildly inaccurate, as many factors vary even when contexts seem 'similar'

Source: adapted from Pagiola et al. (forthcoming)

Table 3.7: Examples of ecosystem disruption and Environmental Health Indicators (EHIs)

Ecosystem	*Service*	*Change*	*Hazard*	*Human health outcome*	*Indicators*
Coastal	Waste processing	Organic overload	Microbes	Diarrhea; Cholera	Incidence
Urban	Cultural?	Air pollution	CO; NO_x; SO_2	Asthma	Morbidity; body burden of metals
Freshwater	Safe water for consumption	Depletion	Poor hygiene	Diarrhea	Childhood mortality
Tropical forest	Water cycle; soil; climate	Deforestation	Infections	Malaria; arbovirus infections	Incidence
Agroecosystem	Food production	Pesticides	Toxic exposure	Reproduction problems	Fertility rates
Freshwater/marine	Provision of fish	Over-harvesting	Depletion of fish resource	Reduced consumption of fish protein	Protein deficiency

Table 3.8: Summary of MA Core Datasets

Core Dataset	*Brief Description*	*Lead agencies*
Global Land Cover	Global Land Cover 2000 (GLC2000) dataset. A global product of land cover in year 2000, based on SPOT Vegetation satellite data.	EU JRC, with regional networks.
Human Population Density	An updated Gridded Population of the World dataset, referenced to year 2000, and including a rural/urban split, including a point database of human settlements >5000 people, an urban mask (polygons), and a complete urban-rural gridded surface.	CIESIN, with World Bank and IFPRI.
Protected Areas	The 13th UN List of Protected Areas, from which a "snapshot" of the extent of Protected Areas in the year 2000 has been generated, as a baseline dataset for the MA.	UNEP-WCMC, with WCPA.
Subnational Agricultural Statistics	Sub-national time series and single year crop production data including area, production, and yield, available for the globe.	IFPRI, with wider consortium
Climate	i) 0.5-degree dataset of monthly surface climate extending from 1901 to 2000 over global land areas, excluding Antarctica. ii) 10-minute mean monthly surface climate grids for the 1961-1990 period covering a similar area.	University of East Anglia CRU, and University of Oxford, UK.
Human Well-being indicators	Sub-national infant mortality, malnutrition and GDP data. Global data, although malnutrition index only available for the developing world.	CIESIN
Areas of Rapid Land Cover Change	A synthesis of the knowledge of areas affected by rapid land cover change during the last twenty years for various change classes, including deforestation, cropland and pasture expansion, soil degradation and desertification, urban expansion and exceptional fire events.	IGBP/IHDP LUCC, GOFC/GOLD
Global MA Reporting 'Units'	Datasets delineating MA system boundaries (see {table 3.9}), biomes and biogeographical realms, and socioeconomic regional reporting units.	Various.

Table 3.9: MA System boundary definitions

MA System	*Description*
Coastal	The area between the interpolated 50 m bathymetry and 50 m elevation contours from the ETOPO2 dataset. The 50 m inland contour is constrained to a maximum distance of 100 km.
Cultivated	Agricultural classes from version 2 of the Global Land Cover Characteristics Dataset (GLCCD v2.0, USGS/EDC 2000). Cropland, pasture and mosaic (or mixed) agriculture and other land use classes are included.
Dryland	A subset of the aridity zone map published in the World Atlas of Desertification. Aridity zones are derived from an Aridity Index (AI) calculated as the ratio of precipitation to potential evapotranspiration. The zones hyperarid, arid, semiarid and dry subhumid are included in the dryland system.
Forest and woodland	Derived from the Global Land Cover 2000 Dataset (GLC2000). Extracted classes are broadleaved, needle-leaved, mixed tree cover, regularly flooded (such as mangroves) and burnt tree cover, and a mosaic tree cover/other natural vegetation class (classes 1 to 10 of the global classification).
Inland water	Includes major rivers, wetlands, lakes, and reservoirs as compiled in the Global Lakes and Wetlands Database-Level 3 (GLWD-3).
Island	Oceanic and coastal Islands as defined by ESRI's ArcWorld Country Boundaries dataset. Approximately 11,925 islands are represented and include those listed as members of the Alliance of Small Island States (AOSIS) and the Small Island Developing States Network (SIDSnet).
Marine	The marine system boundary is defined from the interpolated 50 m bathymetry (from the ETOPO2 dataset) seaward. Longhurst's biome classification provides sub-system categorisations.
Mountain	Derived from UNEP-WCMC's mountain dataset, using criteria of altitude, slope and local elevation range. Altitudinal life zones form sub-system reporting units.
Polar	Arctic and sub-arctic vegetation types define the northern hemisphere portion of the polar system. Vegetation types are delineated from a combination of global and regional land cover maps from remote imagery. Antarctica forms the southern portion of the polar system.
Urban	Derived from the Global Land Cover 2000 Dataset (GLC2000) artificial surfaces class (class 22 in the global legend).

Table 3.10: Data handling procedures in the MA

Data Application in the MA	*Data Handling Procedures*
1. Peer reviewed or validated datasets cited in MA reports	• Full citation in MA report
2. Peer reviewed or validated datasets used in MA analysis (e.g., to calculate area, quantity), map or table but unmodified.	• Full citation in MA report • Included in MA Data Catalog • May be included in datasets available for on-line access as part of MA outreach
3. Non-peer-reviewed datasets cited in MA reports.	• Dataset critically assessed. Quality and validity of the dataset reviewed by chapter team before incorporating results from the source into an MA Report. • The following materials sent to the Working Group Technical Support Unit: Title of dataset; Location (URL if available); Institution responsible for maintaining the data; Information on the availability of the data to other researchers; Contact details for 1-2 people who can be contacted for further information about the source.
4. Non-peer-reviewed datasets used in MA analysis, map or table but unmodified.	• Procedures in category 3 followed. • Included in MA Data Catalog • Included in MA Data Archive if possible (particularly if a key dataset for the analysis) • May be included in datasets available for on-line access as part of MA outreach
5. Data modified in an MA analysis or new datasets produced through existing peer-reviewed data. Considered an "MA Dataset"	• Dataset critically assessed. Quality and validity of the dataset reviewed by chapter team before incorporating results from the source into an MA Report. • MA Metadata Standards followed • Included in MA Data Catalog and MA Data Archive • Made freely available to other users
6. MA Core Datasets	• MA Metadata Standards followed • Included in MA Data Catalog and Data Archive • Made freely available to other users
7. MA Heritage Datasets Datasets representing a valuable 'baseline' condition for Yr 2000 (e.g., NDVI data).	• MA Metadata Standards followed • Included in MA Data Catalog and MA Data Archive • Made freely available to other users

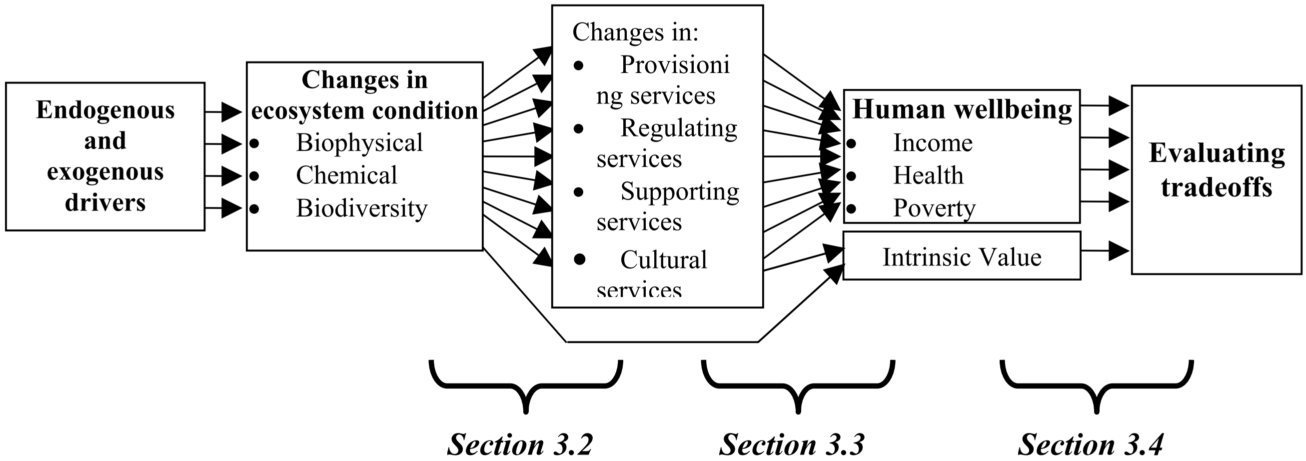

Figure 3.1:Linking ecosystem condition to well-being requires assessing ecosystem condition and its effect on services (section 3.2), the resulting impact on human well-being and other forms of value (section 3.3), and evaluating trade-offs among objectives (section 3.4).

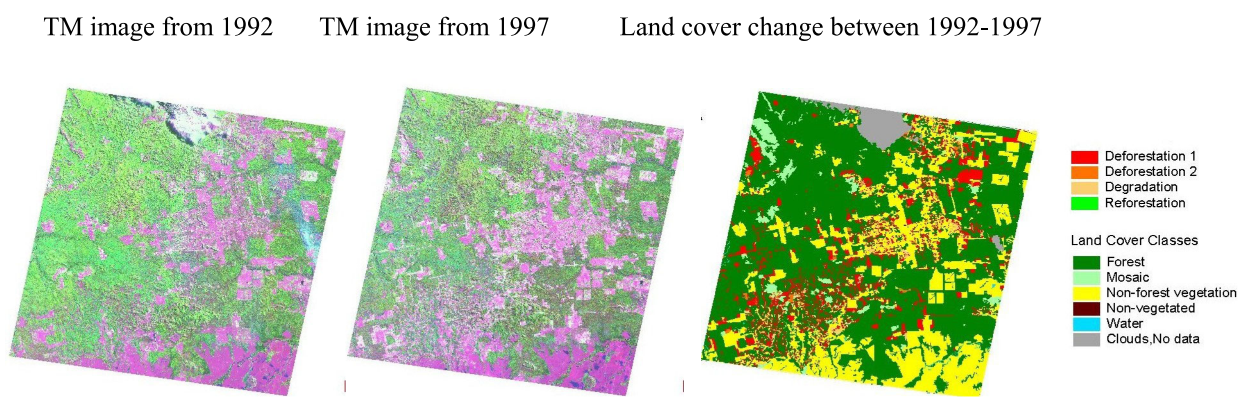

Figure 3.2:Landsat TM scene from 1992, 1997, and land cover change. The scene covers approximately 185 x 185 km^2 in Mato Grosso, Brazil. –add text---

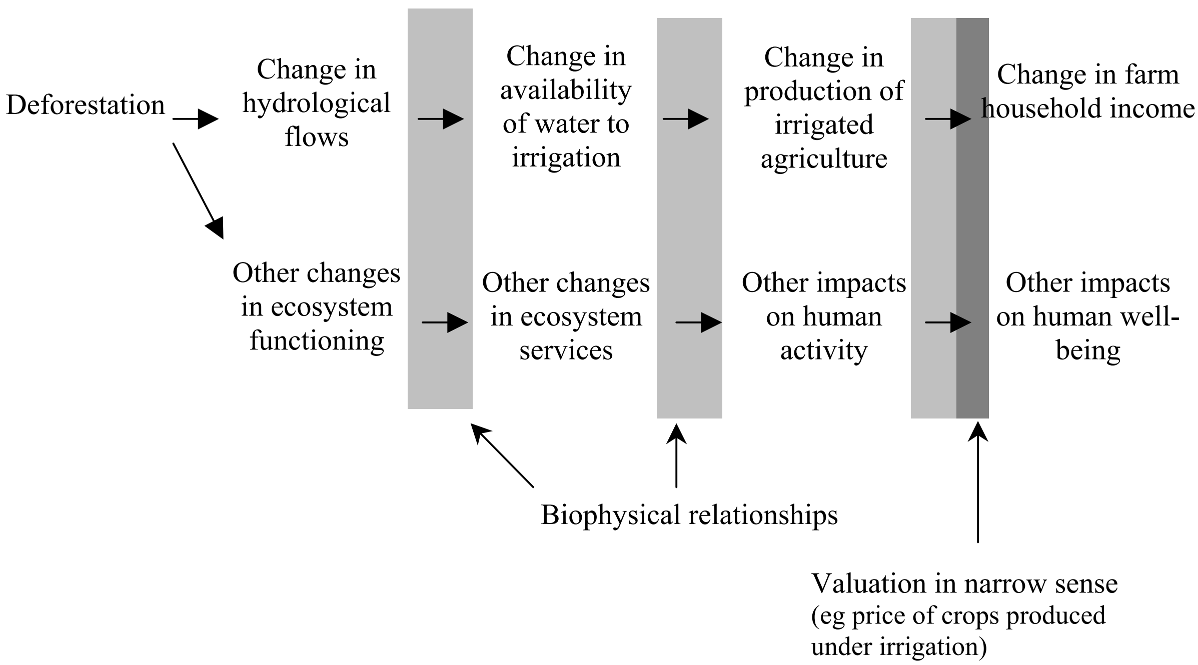

Figure 3.3: Valuing the impact of ecosystem change. Adapted from Pagiola et al. (forthcoming).

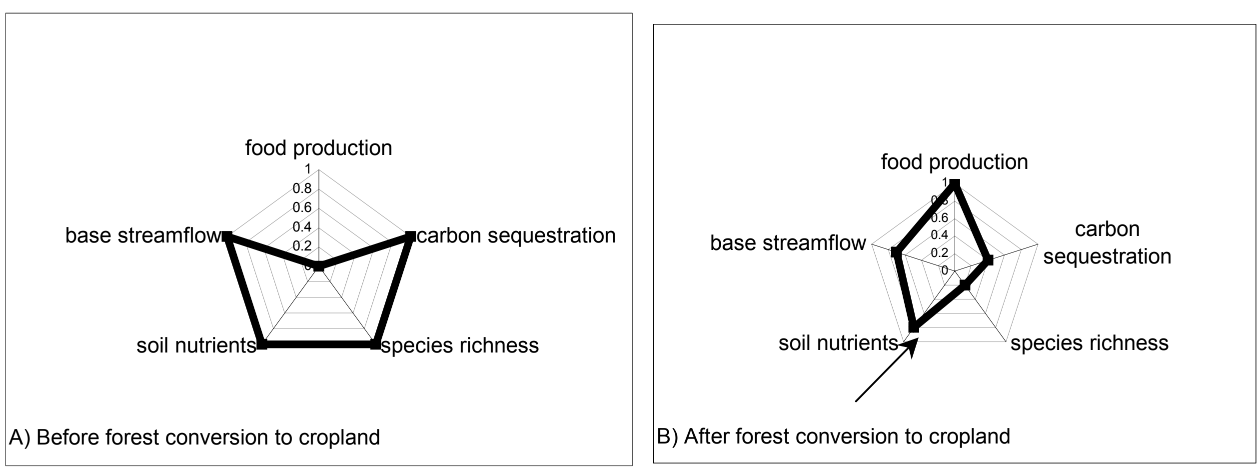

Figure 3.4: Hypothetical trade-offs in a policy decision to expand cropland in a forested area. Indicators range from 0 to 1 for low to high value of service. The values of the indicators vary according to the spatial and temporal scales of interest.

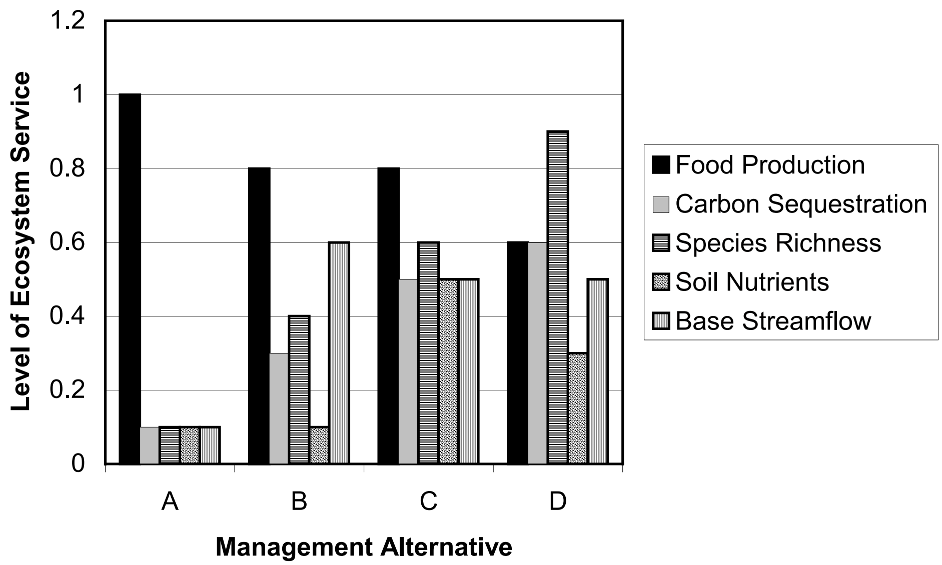

Figure 3.5: Portrayal of hypothetical trade-offs in ecosystem services associated with management alternatives for expanding cropland in a forested area. Indicators range from 0 to 1 for low to high value of service. See text for management alternatives. Adapted from Heal et al. (2001b).

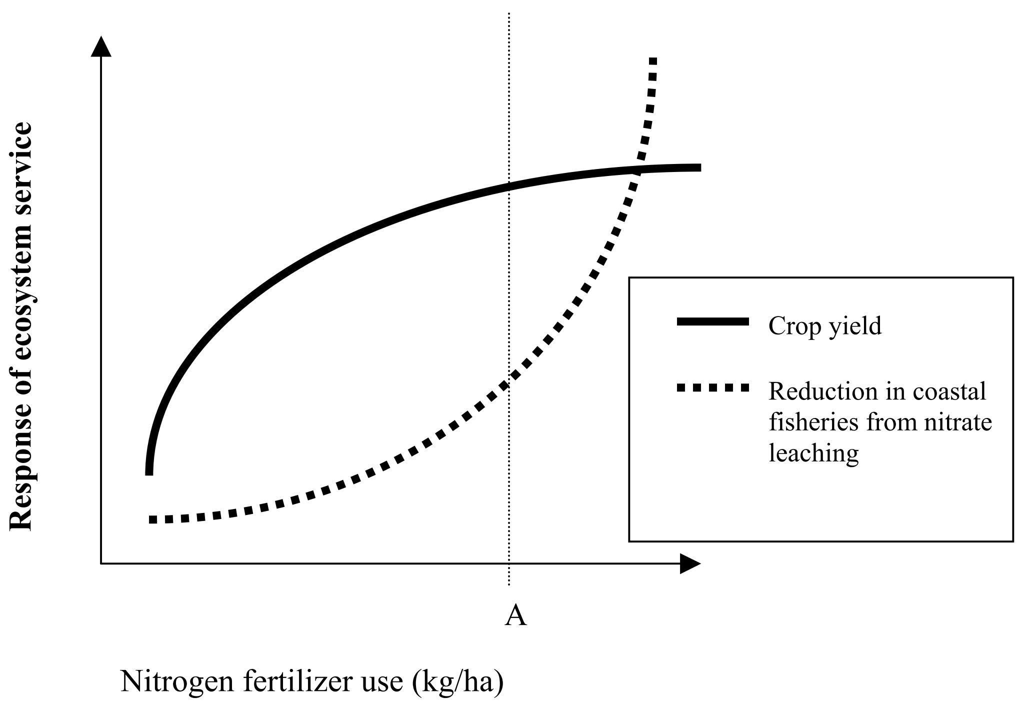

Figure 3.6: Example of non-linear responses of two ecosystem services (crop yields and coastal fisheries) to application of nitrogen fertilizer.

Box 3.1: Criteria for effective ecological indicators

- Does the indicator provide information about changes in important processes?
- Is the indicator sensitive enough to detect important changes but not so sensitive that signals are masked by natural variability?
- Can the indicator detect changes at the appropriate temporal and spatial scale without being overwhelmed by variability?
- Is the indicator based on well-understood and generally accepted conceptual models of the system to which it is applied?
- Are reliable data available to assess trends and is data collection a relatively straightforward process?
- Are monitoring systems in place for the underlying data needed to calculate the indicator?
- Can policymakers easily understand the indicator?

Source: NRC, 2000

Box 3.2: Indicators of biodiversity

The following is a sample of the types of indicators that can be used to monitor status and trends in biodiversity. The list is not exhaustive, and specific choice of indicators will depend on particular scale and goals of the monitoring program.

- *Threatened species*: the number of species that are in decline, or otherwise classified as under threat of local or global extinction (see also {section 3.2.4})
- *Indicator species:* species that can be shown to represent the status or diversity of other species in the same ecosystem. Indicator species have been explored as proxies for everything from whole ecosystem restoration, e.g. (Carignan and Villard 2002), to overall species richness, e.g. (MacNally and Fleishman 2002). The phrase "indicator species" is also used broadly, to include several of the other categories listed here.
- *Umbrella species:* species whose conservation is expected to confer protection of other species in the same ecosystem (e.g., species with large area requirements). If these species persist, it is assumed that others persist as well (Roberge and Angelstam 2004).
- *Phylogenetic or taxonomic diversity:* the number of species weighted by their evolutionary distinctiveness (Mace et al. 2003). This indicator is increased with both high species richness and high levels of taxonomic diversity among species.
- *Endemism:* the number of species found only in the specific area, e.g. (Ricketts in press). Note that this is a scale dependent measure: as the area assessed increases, higher levels of endemism will result.
- *Ecological role:* species with particular ecological roles, e.g. pollinators, top predators, e.g. (Kremen et al. 2002).
- *Sensitive or sentinel species:* trends in species that react to changes in the environment before other species, especially changes due to human activities, e.g. (de Freitas Rebelo et al. 2003). Similarly to the famous "canary in the coal mine," monitoring these sensitive species is thought to provide early warning of ecosystem disruption.
- *Aggregate indicators*: indices that combine information about trends in multiple species, e.g. the Living Planet Index which aggregates trends in species abundances in forest, freshwater, and marine species (Loh 2002) and the Index of Biotic Integrity which combines measures of abundances of different taxa in aquatic systems (Karr and Dudley 1981)